Recharge to a semi-arid, heterogeneous coastal aquifer: Uley South Basin, South Australia

submitted by

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“But I have an idea that what’s written is written with all the glaring defects: and if I’d tried to deliver everything I had in mind, the result might be even more incoherent than it is.” Bruce Chatwin

Dedicated to

My late grandma Isaura who is the wisest person I met in my life, and who always inspired me to achieve good things in life through hard work, honesty and intelligence.

My uncle Tójó, who we lost during this period. Tójó always encouraged the critical spirit in me, and the will to discover new things. He was the person who first spiked in me the curiosity for the Anglo-Saxon world.

May my work honour their legacy and their memory.

Declaration of Originality

I certify that this thesis does not incorporate without acknowledgment any material previously submitted for a degree or diploma in any university; and that to the best of my knowledge and belief it does not contain any material previously published or written by another person except where due reference is made in the text.

Carlos Miraldo Ordens

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Summary

Groundwater is a resource of increasing importance throughout the world, especially in arid and semi-arid regions. Prudent groundwater management is paramount for the sustainability of groundwater systems, both in terms of water quantity and quality. Reliable estimates of groundwater recharge are often a pre-requisite for such purposes, as well as for most groundwater studies. However, groundwater recharge is commonly poorly understood and recharge estimates are usually highly uncertain due to its complicated nature and the lack of data. Distributed groundwater recharge (simply termed ‘recharge’ in what follows) is the vertical downward movement of water through the unsaturated zone, reaching the water table, and going into storage. Recharge can occur through focused and/or diffuse mechanisms. In any assessment of recharge, these mechanisms and other important factors are described by a conceptual model, which serves as the starting point of any recharge characterisation, and is necessary for appropriate selection of recharge estimation methods. Despite the significance of a sound conceptual model of recharge processes, it is often untested in recharge evaluations.

This study explores the recharge processes within the coastal, semi-arid Uley South Basin (USB), Eyre Peninsula, South Australia, and attempts to quantify the spatial and temporal variability in recharge fluxes to the system. This aquifer presents significant management challenges, because it supplies around 70% of the Eyre Peninsula’s water demand, and yet there have been historical declines in groundwater levels approaching mean sea level in places. At the time of this study, USB was managed entirely based on recharge estimates, and reliable recharge estimates remain central to the sustainable allocation of pumping from the basin. A predictive tool capable of simulating recharge across the basin is required, partly for direct management applications, but also to underpin proposed groundwater models of USB.

The carbonate terrain of USB forms a recharge environment that is especially challenging to characterise, and previous studies that have attempted to quantify USB recharge produced a wide range of basin- and time-averaged estimates (i.e. 40

to 200 mm/year). There is a need to seek plausible explanations for the lack of agreement across these studies, particularly because management requires a narrow range of uncertainty in USB recharge. Consequently, the focus of this study is to develop an improved characterisation of USB recharge, and to critically examine field-based and modelling approaches as they apply to the USB conditions. Although the investigation focuses on particular site conditions through a case study, they intend to address general research questions of relevance to many aquifers around the world. That is, guidance is offered on the development of conceptual models, and for critically combining field-based and modelling approaches of recharge estimation, especially for real-world case studies where available data are somewhat limited.

This study's first objective was to develop a conceptual understanding of the recharge mechanisms in USB using mainly existing field data. This allowed for an assessment of traditional field-based recharge estimation techniques, the groundwater chloride mass balance (CMB) and water-table fluctuation (WTF) approaches. These were critically examined as they apply to the USB conditions, and subsequently, adaptations to both methods were proposed to account for local factors. Firstly, the application of the CMB method was modified to account for (i) the spatial distribution of atmospheric chloride deposition, which decreases exponentially with distance from the coast; and (ii) up-gradient recharge areas for each well, which were approximated from chlorofluorocarbons (CFC) age dating and aquifer hydraulic properties. This provided a narrow range of temporally and spatially averaged recharge rates (53–70 mm/year), as well as a preliminary indication of the spatial distribution of recharge across the basin. Secondly, the WTF method was modified to account for pumping seasonality, which resulted in a relatively wide range of temporally and spatially averaged recharge rates (47–128 mm/year), reflecting the large uncertainty in specific yield across the basin. The primary contribution to the USB recharge characterisation from the WTF analysis was the valuable insights into the timing of recharge.

A rigorous assessment of rainfall and groundwater hydrochemistry and isotopic datasets allowed for an improved characterisation of USB recharge mechanisms. Despite that there is no runoff to the sea and that runoff is ephemeral and only persists for tens to hundreds of metres, preferential flow features seem to transmit water deeper into the unsaturated zone rather than to the water table, which is indicated by the differences in rainfall and groundwater chloride concentrations. Chloride and ^{18}O data suggest that a substantial proportion of rainfall occurring in dryer months may be completely evaporated at the surface, and that unsaturated zone water and groundwater are subject to transpiration more so than evaporation. Chloride and bromide rainfall and groundwater data seem to confirm that rainfall in USB is essentially evaporated seawater and that rainfall is the only source of recharge.

The second objective was to investigate the influence of variants of the conceptualisation of recharge processes in USB on recharge predictions based on one-dimensional (1D) unsaturated flow modelling. The study focussed particularly on different complexities of the unsaturated zone lithology and representations of preferential flow. A modified form of the code LEACHM was applied that included a simple representation of preferential flow, whereby runoff was redistributed within predefined regions of the unsaturated zone or bypassed the unsaturated zone, to allow testing of the effects of sinkholes and other preferential flow features. The model outcomes were tested against field-based timings of recharge, which indicated that only the models with preferential flow correctly reproduced the WTF-inferred timing of recharge, and that preferential flow probably redistributes runoff into the unsaturated zone rather than passing it to the water table directly. It was found that vegetation exerts the most significant control on simulated USB recharge, and a better field characterisation of vegetation parameters and distribution would be expected to reduce considerably the recharge modelling uncertainty. Because different but equally plausible conceptual models produce widely varying recharge rates, field-based recharge estimates were shown to be essential to constrain the modelling results.

The third objective was to allow for a comparison between recharge from field-based and modelling methodologies integrated across the basin. This also provided total USB recharge influxes, for later comparison to pumping and other basin-wide fluxes. Temporally and spatially averaged modelled recharge rates were in the range estimated using CMB, and consistency between modelled and fields-based timing of recharge was obtained in the simulations where surface runoff was distributed deeper into the unsaturated zone. The simulations that better matched the field-based estimations produced temporally and spatially averaged recharge rates of 69 and 74 mm/year. Modelling provided an independently-verified fine resolution of recharge distribution in both time and space domains for the basin, which are especially valuable for management purposes and for input to groundwater flow models.

The fourth objective was to evaluate two different groundwater management strategies, the flux-based management (FBM) and the trigger level management (TLM) approaches, as they apply to USB. A simple basin water balance modelling approach was used, which required transient recharge estimates from the modelling efforts. The results indicate that the addition of TLM to the presently used FBM of the system leads to (i) enhanced water availability manifested as higher allowable pumping volumes and fewer zero-pumping months; (ii) reduction in the risk of aquifer degradation and protection against recharge estimate inaccuracies; and (iii) enhanced understanding of basin functioning leading to adaptive management.

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Further acknowledgements, directly related to each submitted paper, are included at the end of each chapter, in the same form as they were listed in each of the journal paper.

1 Introduction

1.1 Scope

Groundwater is a resource of crucial importance worldwide. An increasing number of people are coming to depend on groundwater as a source of freshwater, especially in arid and semi-arid regions (Healy, 2010). Effective management is essential to ensure the longevity of groundwater resources and to preserve its quality. Management and planning strategies require estimates of groundwater recharge (Simmers, 1990). Despite being an essential element in many groundwater studies, recharge is often poorly understood and challenging to quantify due to difficulties in accounting for spatial and temporal variability, and in directly measuring recharge (Healy, 2010).

The current study focuses on recharge derived directly from rainfall, which is the vertical downward movement of infiltrated rainwater through the unsaturated zone, reaching the water table and adding to groundwater storage (e.g. Scanlon et al., 2002a). Other recharge processes include inter-aquifer exchanges (whereby one aquifer is recharged from the discharge of another), inflows from flooding, surface water-groundwater interactions, managed aquifer recharge associated with aquifer storage and recovery systems, and the intrusion of seawater arising from lowering coastal aquifer water levels. Recharge can be diffuse or focused. Diffuse recharge occurs throughout a large area, typically originating from rainfall infiltration and percolation through the unsaturated zone. Focused recharge is concentrated in smaller areas, and originates from surface water bodies (e.g. lakes, rivers and streams) or preferential flow features (e.g. sinkholes, playas and cracks).

The existing literature on groundwater recharge is vast. It includes textbooks (e.g. Lerner et al., 1990; Healy, 2010), review articles (e.g. Scanlon et al., 2002a; Healy and Cook, 2002), and publications documenting field-based studies (e.g. Allison and Hughes, 1978) and recharge modelling (e.g. Keese et al., 2005). Nevertheless, Healy (2010) stresses that many gaps still remain in understanding recharge processes, applying estimation techniques, obtaining reliable recharge rates, etc. In particular, Healy (2010) highlights difficulties in (i) developing correct conceptual models, (ii) analysing spatial/temporal scales of recharge, and (iii) obtaining data, as the main challenges in recharge studies.

Estimating recharge and identifying recharge processes are difficult due to a number of factors. These include the complex and multifaceted nature of recharge processes (e.g. preferential flow, influence of vegetation, seasonality of climatic conditions, etc.), spatial and temporal variability in recharge rates, and both scarcity and uncertainty in measurements. A sound conceptual model of recharge mechanisms is vital to the success of any recharge study (e.g. Healy, 2010). This is based on existing data and provides insights into (i) processes affecting recharge, (ii) the most suitable methods for recharge estimation, and (iii) the basic elements of the unsaturated zone water balance (in some cases leading to preliminary quantitative recharge estimates). Healy (2010) argues that the most serious sources of errors and unreliability in recharge studies are commonly related to erroneous conceptual models.

Numerous recharge quantification approaches are available. The accuracy of each approach is dependent on a multitude of factors, including the site conditions (e.g. geology, hydrogeology, climate, etc.) and data availability (Scanlon et al., 2002a). Recharge estimates inferred from field measurements apply to various temporal and spatial scales. Point estimates of recharge may be obtained from soil testing (e.g. lysimeter measurements and soil chloride analyses), whereas groundwater-based methods such as the saturated zone chloride mass balance (CMB) method, chlorofluorocarbon (CFC) dating and water-table fluctuation (WTF) analysis produce spatially averaged estimates (Allison and Hughes, 1978; Kitching and Shearer, 1982; Gee et al., 1992; Healy and Cook, 2002; Scanlon et al., 2002a). Lysimeters and the WTF approach offer insights into the temporal variability in recharge, whereas most other methods produce long-term averages.

There can be considerable disagreement and sometimes bias between recharge estimation methods. For example, based on a review of 172 recharge studies across

Australia, Crosbie et al. (2010) found that WTF estimates were higher than those based on the CMB method. They attributed this to the difference in timeframe over which the methods operate (i.e. whereby CMB represents pre-development recharge) and to the potential contribution of transpiration from the saturated zone (accounted for in the CMB method, but not in the WTF approach). Difficulties in the application of methods such as WTF and CMB have been reported in several studies. For example, the accuracy of WTF results depends on factors such as the determination of specific yield, time-variant pumping rates and diffusivity of the aquifer (e.g. Healy and Cook, 2002; Moon et al., 2004; Crosbie et al., 2005; Cuthbert, 2010). The primary challenge of applying the saturated CMB approach relates to the quantification of the chloride input (e.g. Scanlon et al., 2006), which can be influenced by distance to the coast (e.g. Hutton, 1976), vegetation (e.g. Beier and Gundersen, 1989), irrigation (e.g. Scanlon et al., 2010) and rainfall variability.

Scanlon et al. (2002a) advocate the use and comparison of different methods for quantifying groundwater recharge. The application of multiple methods does not guarantee more accurate quantification of recharge, but will highlight errors in the form of inconsistent outcomes (Healy and Cook, 2002). These errors may stem from the use of methods beyond their limits of validity or from invalid assumptions about the nature of the recharge processes, in addition to other limiting factors, such as parameter uncertainty and measurement errors.

Keese et al. (2005) support the application of one-dimensional (1D) unsaturated zone modelling to calculate recharge. It is the only approach capable of predicting future recharge patterns, while enabling examination of the relative importance of various controlling factors. They highlight that the primary difficulties in recharge modelling are data availability and the selection of model parameters, especially at regional scales. Similarly, Scanlon et al. (2002a) suggest that the range of recharge rates obtained using numerical modelling can vary greatly due to difficulties in characterising hydraulic conductivity and nonlinearities between hydraulic conductivity, water content, and matric potential. Other factors have been identified as sources of uncertainty in 1D recharge modelling, such as boundary conditions (e.g. Carrera-Hernández et al., 2012), profile discretisation (e.g. Carrera-Hernández et al., 2012), model conceptualisation (e.g. Scanlon et al., 2003) and preferential flow (e.g. Köhne et al., 2009). These examples emphasise the inherent non-uniqueness that arises in 1D recharge models. As such, models need to be substantiated against alternative recharge estimation techniques, because deterministic recharge modelling without calibration is known to produce highly uncertain estimates.

Previous studies have attempted to address model complexities by exploring the sensitivities of recharge predictions to changes in model inputs, thereby providing insight into the uncertainty of model predictions, as well as guidance on data collection (e.g. Scanlon et al., 2003). For example, Keese et al. (2005) studied the factors that control recharge across the state of Texas (USA). A sensitivity analysis was undertaken to examine the effect of variations of vegetation, unsaturated zone texture, climatic inputs and profile depth on predicted recharge. Their models were particularly sensitive to vegetation characteristics, potential evapotranspiration rates and the unsaturated zone texture. Smerdon et al. (2008) also explored the sensitivity of 1D recharge models for shallow glacial outwash aquifers in Canada. They concluded that recharge was mainly influenced by climate and the depth to water table (DWT). Scanlon et al. (2010) found that the vegetation rooting depth was a sensitive parameter in obtaining simulated recharge values that were consistent with field estimates in the Thar Desert in India. From these studies, it follows that the most sensitive parameters and processes in recharge models may depend on the physical characteristics of the system. As such, the uncertainty in recharge estimation techniques is most likely site specific, particularly considering that complex processes such as preferential flow, which exert a strong control on recharge in some settings (e.g. Ireson and Butler, 2011), are represented to varying degrees (or often not simulated) in recharge investigations. This is particularly the case for regional-scale studies (e.g. Keese et al., 2005).

There are fundamental differences between modelling and field-based approaches to the estimation of groundwater recharge. Unsaturated zone modelling uses atmospheric driving forces in simulating key atmosphere-vegetation-unsaturated zone interactions, thereby predicting recharge from the combination of several biophysical processes. The basic premise to modelling approaches is that rainfall is segregated into its various constituents, including canopy interception, runoff, rapid infiltration through preferential pathways, diffuse infiltration through the soil matrix, evaporation at the soil surface, evapotranspiration within the soil profile, storage within the vadose zone, and recharge to the water table. In some cases lateral movements in unsaturated zone flow may also be considered (e.g. Köhne et al., 2009). In contrast to 1D modelling, field-based approaches usually infer integrated recharge responses by measuring hydrological and hydrochemical parameters at various points in the water cycle. For example, groundwater level observations may be used to interpret recharge according to the response of the groundwater system to recharge influxes (i.e. the WTF method), albeit this approach can fail to properly account for both recharge and discharge stresses from various sources/sinks within the catchment (Healy and Cook, 2002).

Despite the uncertainty associated with recharge modelling, results are often not verified against field-based recharge estimates (e.g. Jyrkama et al., 2002). A few exceptions can be found in the literature where results from both methodologies are compared (e.g. Keese et al., 2005; Anuraga et al., 2006; Scanlon et al., 2010; Ireson and Butler, 2011). The comparisons between approaches by Keese et al. (2005), Anaruga et al. (2006) and Scanlon et al. (2010) considered temporally averaged recharge rates, i.e., they focused on the spatial variability of recharge. In all three cases, the match between the 1D unsaturated zone modelling approach and CMB estimates was mixed. For example, Keese et al. (2005) reported that modelling results fall in the range of the CMB estimates in only three out of thirteen regions of their study area. In four other regions, the modelling predictions were around 72% lower on average than the field-based estimates (the comparison between modelling results and field data was not reported for the remaining regions). They explained the discrepancies between the two approaches as possibly a result of (i) focused recharge (originated from playas) occurring in two of the regions that could not be captured by the 1D model (i.e. only matrix flow was considered), and (ii) field measurement bias towards units of higher permeability. Ireson and Butler (2011) compared 1D modelled recharge rates (considering preferential flow) with water-table fluctuation data in Chalk aquifers in the Pang and Lambourn catchments (UK). They found that the modelled timing of recharge was reasonably consistent with water-table fluctuations, although lags between modelled recharge pulses and water-table rises were observed. They speculated that the differences were attributable to lateral water movements within the saturated zone. Further modelling was suggested as a means to decipher the cause of differences in timing between the observed water levels and 1D modelling approaches.

The foregoing examples reveal an overall lack of agreement between modelled and field-based recharge estimates. There are few critical evaluations of discrepancies between field-based and modelling methods. Even when recharge time-averages are in agreement between approaches, often no consideration is given to the consistency between modelled and observed timings of recharge. There is a need for further research into the causes of discrepancies between recharge estimation approaches applied to basin-scale investigations, and for an evaluation of commonly applied methods of estimation to better combine the various field- and modelling-based approaches.

1.2 Study area

This body of research uses the coastal, topographically closed Uley South Basin (USB), Eyre Peninsula, South Australia, as a case study. Abstraction of groundwater from this sedimentary basin supplies around 70% of the water demand of the Eyre Peninsula's population of around 33,500 (Zulfic et al., 2007), with the remainder being met by surface water sources or by groundwater from other basins. As with many arid and semi-arid regions, there are limited alternative water supply options in the Eyre Peninsula, and therefore recharge estimates are crucial. At the time of this study, management of groundwater abstraction in USB was based entirely on recharge estimates. Both recharge variability and long-term averages of recharge underpin water management decision-making. Given historical declines (from the 1970s until the 2000s) in groundwater levels within the basin (EPNRM, 2006), there is an urgent need to re-evaluate recharge to the basin. The evaluation of management practices, climate variability and land-use impacts requires a predictive tool that is capable of simulating USB recharge, accounting for its spatial and temporal variability.

The carbonate terrain of USB forms a recharge environment that is especially challenging to characterise. Considerable variations in land cover, topography, water-table depth and soil type are encountered across the basin, and it has been suggested that dissolution features create complicated recharge pathways in the basin, which thereby receives both diffuse and focused infiltration (Evans 1997). Previous studies have attempted to quantify recharge in USB using various methods (e.g. WTF, CFC, CMB and water balance analysis), producing a wide range of basin-averaged and time-averaged estimates (i.e. 40 mm/year to 200 mm/year; Morton and Steel, 1968; Sibenaler, 1976; Barnett, 1978; EWS, 1984; Evans, 1997). Although there is considerable disagreement amongst previous studies, there has been no attempt to reconcile differences and seek to understand the causal factors for any bias in the different approaches. Therefore, there is a need to seek explanations for the lack of agreement across previous recharge studies, and to obtain a narrower range of recharge rates for the basin that can be used for management purposes.

The USB case study presents numerous challenges in terms of recharge characterisation due to its physical setting and to data availability. First, there are many conceptual questions about the recharge processes occurring in the basin, such as the role of preferential flow, the importance of heterogeneous lithological composition of the unsaturated zone, and the vegetation effects. Second, the basin shows high spatial variability in conditions likely to impact recharge, namely vegetation types, substrate characteristics, water-table depth and topographical slope. Third, there is a scarcity of groundwater data (e.g. groundwater and rainfall isotopic and chemical compositions, and water levels) to better conceptualise the system and to estimate recharge based on field-based methods. Fourth, due to the nature of the lithology, it is difficult to obtain unsaturated zone data for the development of mathematical models of the unsaturated zone. Fifth, there are questions regarding the applicability of field-based methods (e.g. CMB, WTF and CFC dating) without taking into account some site-specific factors. These include the effect of vegetation and proximity to the coast on the atmospheric chloride deposition, seasonality of pumping, and the mixing of environmental indicators of recharge in the aquifer. Because obtaining a spatial and temporal fine resolution of recharge to USB is paramount for management purposes, this basin provides an opportunity for developing and implementing an approach that explores conceptualisation questions and reconciles different recharge estimation approaches in a context of a data scarcity.

1.3 Objectives

This study aims to explore recharge and accompanying hydrological processes to critically examine field-based and modelling approaches as they apply to specific site conditions. Although the investigation focuses on particular site conditions through a case study, they intend to address general research questions of relevance to many aquifers around the world. That is, guidance is offered on the development of conceptual models, and for critically combining field-based and modelling approaches of recharge estimation, especially for real-world case studies where available data are somewhat limited.

The specific objectives are as follows. The first objective is to develop a conceptual understanding of the recharge mechanisms in USB using hydrogeologic and water quality observations, leading to quantification of the basin’s recharge using the CMB and WTF approaches. Because of the wide range of previously estimated recharge rates (Harrington et al., 2006; EPNRM, 2009), the applicability of the CMB and WTF approaches under the conditions encountered in USB are critically examined. Adaptations of both methods are proposed to account for local factors. The second objective of the study is to investigate the influence of variants of the conceptualisation of recharge processes in USB on 1D modelling predictions. The study focused on two aspects of model conceptualisation: (i) the complexity of the unsaturated zone lithology, and (ii) runoff infiltration and preferential flow (matrix flow versus bypass flow). The model outcomes were tested against the timing of recharge derived from the WTF method. The third objective of study is to combine 1D modelling and field-based methods, using a Geographical Information System (GIS), to extend recharge predictions to the basin scale, and thereby to estimate spatial and temporal distributions of USB recharge that are improvements on previous attempts. The combination of field and modelling techniques is critically evaluated. The fourth objective is to apply modelled recharge rates to a simple lumped-parameter model of the basin's water balance, which is used to evaluate different groundwater management practices in USB. The findings are used in combination with "lessons learnt" from other published accounts of coastal aquifer management strategies to offer a reflective assessment of management practices for regulated coastal groundwater systems.

1.4 Outline of subsequent chapters

Each of the following chapters corresponds to journal papers (Chapters 2, 3 and 4), or extracts from these (Chapter 5) (reflecting the student's contribution where he was not the principal author of the publication). Chapter 2 offers a description of the conceptual model of the recharge processes in USB using available field data, and provides field-based recharge estimates based on adapted versions of CMB and WTF. Chapter 3 investigates the effect of different conceptualisations on 1D recharge modelling, as they apply to USB conditions, and offers guidance into the likelihood of such conceptual models through comparison of results with field-based estimates. Chapter 4 applies findings from Chapters 2 and 3 to obtain spatially and temporally variable recharge rates at the basin-scale through 1D modelling linked to GIS. Chapter 5 is adapted from Werner et al. (2011), where preliminary modelled recharge rates by Ordens et al. (2010) were used to test different groundwater management approaches in USB, through a simple lumped-parameter water balance model of the basin. Finally, Chapter 6 encompasses the main conclusions arising from this thesis. The references of the papers are as follows:

Chapter 2: Ordens CM, Werner AD, Post VEA, Hutson JL, Simmons CT, Irvine BM. 2012. Groundwater recharge to a sedimentary aquifer in the topographically closed Uley South Basin, South Australia. *Hydrogeology Journal* 20: 61–72.

Chapter 3: Ordens CM, Post VEA, Werner AD, Hutson JL. 2014. Influence of model conceptualisation on one-dimensional recharge quantification – Uley South, South Australia. *Hydrogeology Journal* 22: 795– 805.

Chapter 4: Ordens CM, Werner AD, Post VEA, Hutson JL, Knowling MK. An application of numerical modelling to extend field-based recharge estimates to the basin scale: Uley South aquifer, South Australia. [In preparation for submission].

Chapter 5: Werner AD, Alcoe DW, Ordens CM, Hutson JL, Ward JD, Simmons CT. 2011. Current practice and future challenges in coastal aquifer management: Flux-based and trigger-level approaches with application to an Australian case study. *Water Resources Management* 25: 1831–1853.

2 Groundwater recharge to a sedimentary aquifer in the topographically closed Uley South Basin, South Australia

Abstract

The chloride mass balance (CMB) and water-table fluctuation (WTF) analysis methods were used to estimate recharge rates in the Uley South Basin, South Australia. Groundwater hydrochemistry and isotope data were used to infer the nature of recharge pathways and evapotranspiration processes. These data indicate that some combination of two plausible processes is occurring: (i) complete evaporation of rainfall occurs, and the precipitated salts are washed down and redissolved when recharge occurs, and (ii) transpiration dominates over evaporation. It is surmised that sinkholes predominantly serve to by-pass the shallow soil zone and redistribute infiltration into the deeper unsaturated zone, rather than transferring rainfall directly to the water table. Chlorofluorocarbon measurements were used in approximating recharge origins to account for coastal proximity effects in the CMB method and pumping seasonality was accounted for in the WTF-based recharge estimates. Best estimates of spatially and temporally averaged recharge rates for the basin are 52–63 mm/year and 47–129 mm/year from the CMB and WTF analyses, respectively. Adaptations of both the CMB and WTF analyses to account for nuances of the system were necessary, demonstrating the need for careful application of these methods.

2.1 Introduction

Reliable estimates of recharge are often a key prerequisite for the proper management of groundwater resources. The measurement of groundwater recharge and the interpretation of its processes are difficult due to a number of factors, including the complex and multifaceted nature of recharge processes, spatial and temporal variability in recharge rates, and both scarcity and uncertainty in measurements pertaining to recharge estimation. For these reasons, Scanlon et al. (2002a) advocate the use and comparison of different methods for quantifying groundwater recharge. The application of multiple methods does not guarantee more reliable outcomes but will highlight errors in the form of inconsistent outcomes (Healy and Cook, 2002). These errors may stem from the use of methods beyond their limits of validity or from invalid assumptions about the nature of the recharge processes. A sound conceptual model of the recharge mechanisms is vital to the successful application of recharge estimation techniques.

A wide variety of recharge quantification approaches is available. The accuracy of different methods is dependent on a multitude of factors, including the site conditions (e.g. geology, hydrogeology, and climate) and data availability (Scanlon et al., 2002a). Recharge estimates inferred from field measurements apply to various temporal and spatial scales. Point estimates of recharge may be obtained from soil testing (e.g. lysimeter measurements and soil chloride analyses), whereas groundwater-based methods such as the chloride mass balance (CMB) method, chlorofluorocarbon (CFC) dating and water-table fluctuation (WTF) analysis produce spatially averaged estimates (Allison and Hughes, 1978; Kitching and Shearer, 1982; Gee et al., 1992; Healy and Cook, 2002; Scanlon et al., 2002a). Lysimeters and the WTF approach offer insight into the temporal variability in recharge, whereas most other methods produce long-term averages. Based on a review of 172 recharge studies across Australia, Crosbie et al. (2010) found that WTF estimates were higher than those based on the CMB method. They attributed this to the difference in timeframe over which the methods operate (i.e. whereby CMB represents pre-development recharge, while WTF represents recharge over the length of time that measurements of water levels were recorded), and to the potential contribution of evapotranspiration from the saturated zone, which is reflected in the CMB results but not necessarily in the WTF results.

The focus of the current study is the characterisation of recharge in the Uley South Basin (USB), Eyre Peninsula, South Australia. Abstraction of groundwater from the basin supplies around 70% of the water demand of the Eyre Peninsula’s population of 33,500 (Werner et al., 2011), with the remainder being met by surface water sources or by groundwater from other basins. Both recharge variability and long-term averages rates of recharge are of high importance for water management decision-making, particularly given historical declines in groundwater levels within the basin (EPNRM, 2006; Werner et al., 2011). The limestone terrain of the USB forms a recharge environment that is especially challenging to characterise. Spatial variations in land cover, topography, water-table depth and soil type, and, it is hypothesised that, the presence of dissolution features all contribute to create a complicated pattern of recharge pathways. Diffuse recharge is expected to occur in areas with undisturbed calcrete and well-developed soils, whereas sinkholes may potentially act as point sources for recharge.

Previous work in other systems characterized by sinkholes has not resulted in a consistent picture of the contribution of sinkholes to recharge on a regional scale. Allison et al. (1985) determined recharge below undisturbed calcrete containing wide-diameter, old (> 40,000 years) sinkholes and small, young (< 100 years) sinkholes based on vertical profiles of chloride (Cl) and isotopes in soil water. They found that the highest recharge rates were associated with the small-diameter sinkholes and that these accounted for most of the regional recharge based on an aerially-weighted average. Wood and Sanford (1995) investigated the recharge processes in a sedimentary aquifer capped by a calcrete layer with sinkholes (referred to as macropores), using Cl and isotopic data in groundwater. One interpretation of the solute and isotopic data indicated that significant sinkhole recharge (i.e. 10% of total aquifer recharge) could be occurring at the regional level. On the other hand, Herczeg et al. (1997) found that localized recharge to a limestone aquifer via sinkholes contributed less than 10% of the total regional recharge and was only important after sustained periods of rainfall. Globally, the role of sinkholes in generating recharge appears to be variable and a general understanding of the influence of sinkholes on recharge remains unresolved.

The primary aim of this study is to develop a conceptual understanding of the recharge mechanisms in USB using hydrogeologic and water quality observations, leading to quantification of the basin's recharge using the CMB and WTF approaches. Previous studies have attempted to quantify USB's recharge using various methods (e.g. WTF, CFC, CMB and water balance analysis), producing a wide range of basin-averaged and time-averaged estimates (i.e. 40 mm/year to 200 mm/year; Harrington et al., 2006; EPNRM, 2009). There is a need to seek plausible explanations for the lack of agreement across previous recharge studies, and therefore the focus of this study is to explore recharge and hydrogeological processes and to critically examine CMB and WTF approaches as they apply to the USB conditions. Adaptations of both methods are proposed to account for local factors that are deemed to be important.

2.2 Study Area

USB (34°47'S, 135°32'E) is located in the southern part of Eyre Peninsula, South Australia (Figure 2.1), and is a topographically closed surface drainage basin. The central part of the basin has a flat to undulating land surface at an elevation of less than 10 m AHD (Australian Height Datum, roughly equal to mean sea level). The basin is bounded to the northwest, northeast and southeast by topographic rises from 50 m AHD up to 170 m AHD, and to the southwest by coastal cliffs of up to 140 m AHD that border the Southern Ocean. The morphology of the basin is associated with ancient dune systems overlying basement ridges and troughs, with distinct dunal landforms and subtle undulations defining local surface drainage systems (Harrington et al., 2006). Surface water systems are highly ephemeral, flowing only after moderate-to-heavy rainfall and persisting for tens to hundreds of metres before terminating abruptly at sinkholes within local surface depressions (Harrington et al., 2006). An important feature of the basin is that there are no surface water outflows to the sea, and therefore at the basin scale, rainfall is partitioned into only evapotranspiration and groundwater recharge. No surface expressions of groundwater are evident in the basin.

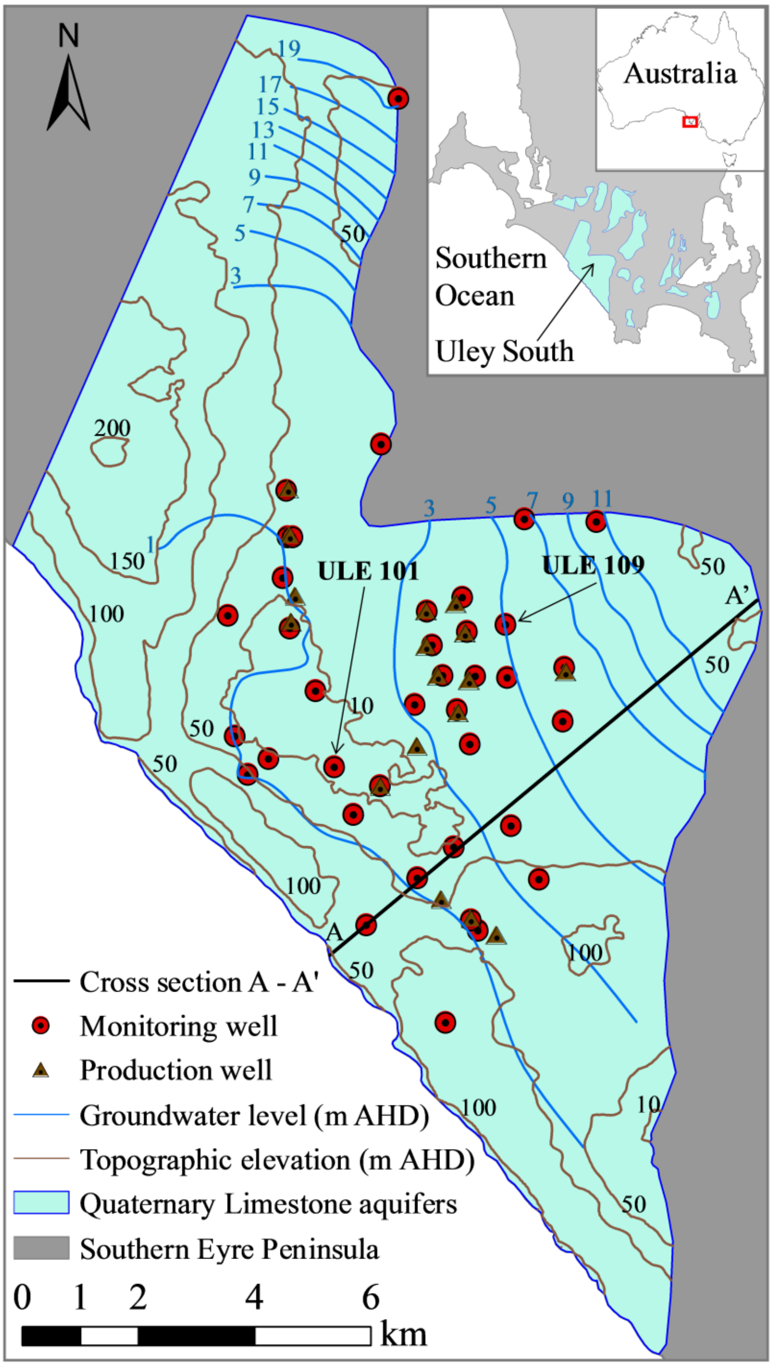

Figure 2.1 – Extent of USB, showing groundwater levels and surface elevations, and the location of monitoring wells (i.e. those used in recharge calculations) and production wells in the Quaternary Limestone aquifer

The climate is semi-arid, characterised by wet winters and dry summers (Zulfic et al., 2007). Climate data for the study area were obtained from the Bureau of Meteorology (2010) for Big Swamp monitoring station (Station no. 18017; located 18 km NE of USB), where daily data are available dating back to 1889. The daily minimum temperatures range from 0 to 26°C, averaging 11°C, while the daily maximum temperatures range from 8 to 45°C, averaging 20°C (for the period 1889-2009; Bureau of Meteorology 2010). The average annual rainfall is 560 mm/year, the average pan evaporation is 1547 mm/year, and the average FAO56 reference evapotranspiration is 1084 mm/year (Bureau of Meteorology 2010).

Figure 2.2 shows the groundwater hydrograph from observation well ULE 101, together with monthly rainfall data at Big Swamp monitoring station. Groundwater levels are measured at approximately monthly intervals in some 42 wells across the basin. A clear seasonal fluctuation in groundwater level can be recognized, together with a steady decline in groundwater levels since the middle of the 1980's. This decline is attributed to the combined effects of pumping and below-average rainfall, and there are presently concerns about movements of the freshwater-seawater interface in the coastal aquifer (Werner et al., 2011).

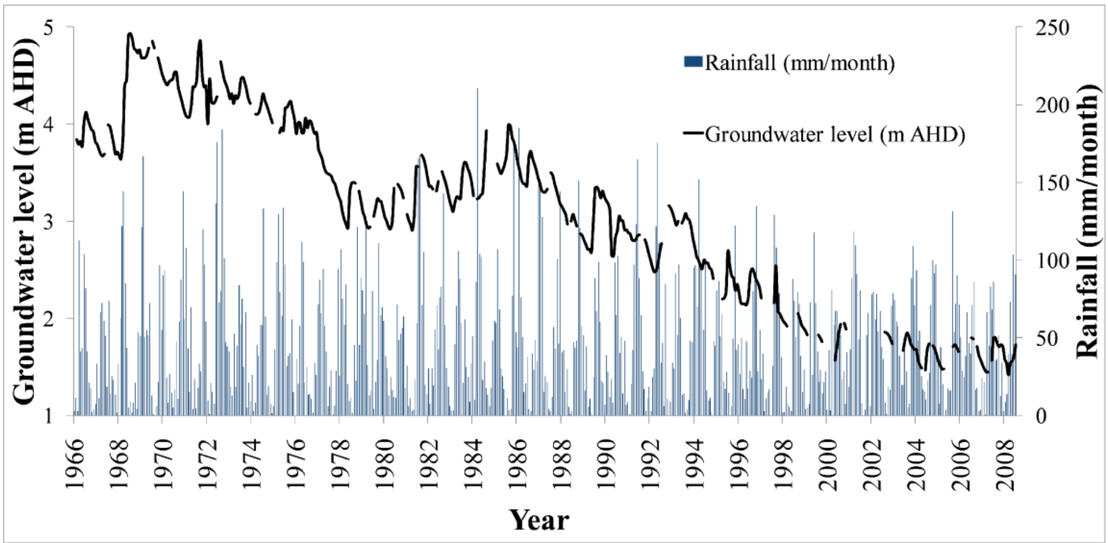

Figure 2.2 – Hydrograph of monitoring well ULE 101 (see Figure 2.1 for its location), and monthly rainfall at Big Swamp monitoring station, modified from Werner et al. (2011)

Soils are typically lithosols (calcareous soils or shallow loamy soils), generally less than 3 cm thick or absent in areas of outcropping surface limestone (calcrete). Shallow depressions may contain up to 5 cm of loamy soils (Harrington et al., 2006). No soils have developed in the modern sand dune areas (designated "deep sand" in Figure 2.3a). The lack or absence of soils across large areas of the basin causes rapid surface runoff, leading to almost instantaneous infiltration via a vast array of sinkholes, which vary in size from millimetres to several metres in diameter. Sinkholes are especially dense in the calcrete and lightly vegetated regions of Figure 2.3a.

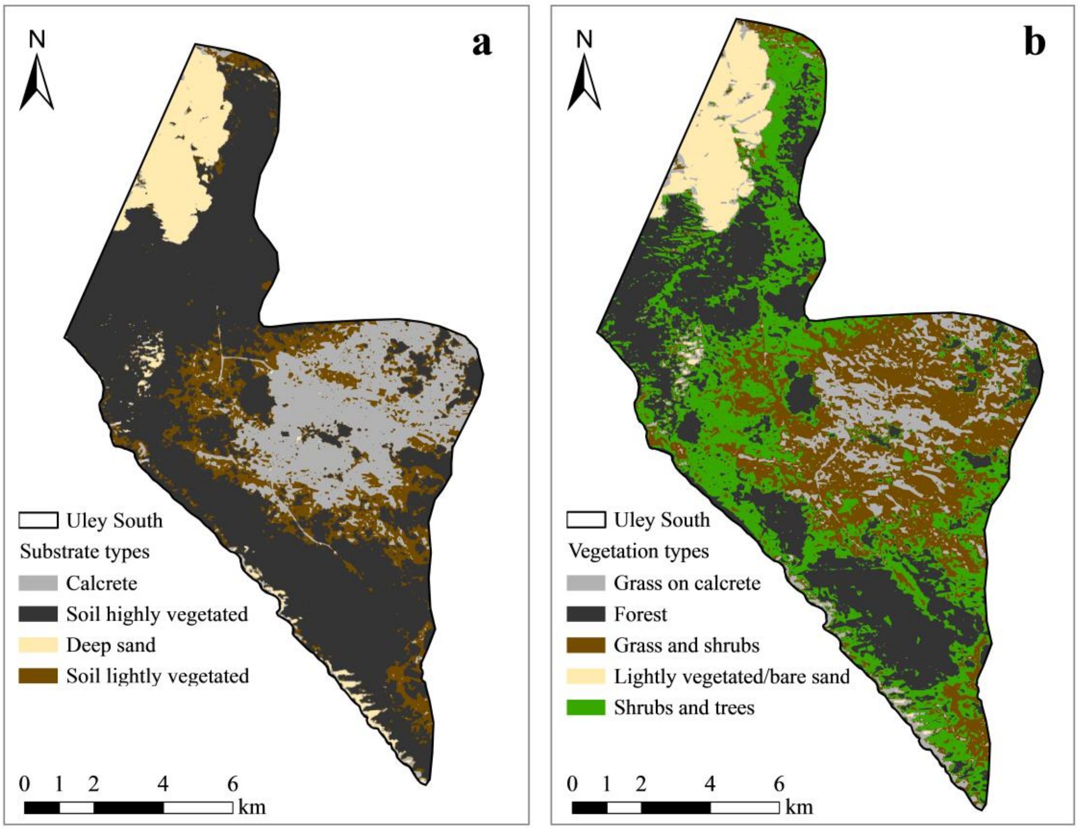

Figure 2.3 – (a) Substrate and (b) vegetation types of USB

Dense stands of mallee vegetation (*Eucalyptus diversifolia* and *Eucalyptus gracilis*) occur above 30 m AHD. Below this level, only isolated stands occur. Some areas are dominated by drooping she-oak (*Allocasuarina verticillata*) that were probably the predominant vegetation species prior to extensive modification most likely from pastoral activities (Harrington et al., 2006). There are large areas of open, sparse grassland, and some infestations of woody weeds (Way, 2006). Remotely sensed multi-spectral images were used to identify the substrate cover (Figure 2.3a) and vegetation types (Figure 2.3b). Landsat Enhanced Thematic Mapper Plus (ETM+) remotely sensed scenes (multi- in the multi-spectral images). The substrate types were classified according to: calcrete, soil lightly vegetated, soil highly vegetated, and sand. Likewise, the vegetation types were classified according to: grass on calcrete, grass and shrubs, shrubs and trees, forest, and lightly vegetated/bare sand (in the sand dune area).

The stratigraphy of the Southern Eyre Peninsula comprises Tertiary and Quaternary sediments unconformably overlying an Achaean metamorphic basement composed of gneiss and quartzites (Harrington et al., 2006). A geological cross section is given in Figure 2.4. The basement geometry shows troughs and ridges with a general NE-SW direction, and the geometry of the sedimentary units that constitute the aquifers reflect the infilling of the basement troughs by Tertiary and Quaternary sediments (Harrington et al., 2006). The Quaternary sediments extend from a thin layer covering the underlying units to extensive accumulations of over 130 m of aeolian fine sands and shell fragments (Evans, 1997). The lower hydrostratigraphic unit (Tertiary Sand) is known as the Wanilla Formation, which consists of fluvial sands, clays and grits with some lignite lenses in the base. The Tertiary Sand aquifer is referred to by Harrington et al. (2006) as confined to unconfined, depending on the presence of a clay layer. This formation has a maximum thickness of 60 m and can be locally absent (Harrington et al., 2006). A lateritic clay sequence (Tertiary Clay) of up to 25 m thick, known as the Uley Formation, is found in the upper Tertiary sediments across much of the basin, although this palaeosol horizon is discontinuous in places (Harrington et al., 2006). Tertiary sediments are overlain by the Quaternary Limestone, also known as the Bridgewater Formation, corresponding to the unconfined Quaternary Limestone aquifer, which is composed of a combination of unconsolidated aeolian sediments and limestone of varying hardness and porosity, presenting karst features in places (Harrington et al., 2006). The thickness of the Quaternary Limestone sequence is over 130 m in areas of high topographic relief such as the coastal limestone cliffs, and the base of the Quaternary aquifer occurs to elevations of -60 to 50 m AHD (Zulfic et al., 2007; Fitzpatrick et al., 2009). A surface or near-surface calcrete horizon is extensive across the basin, along with secondary porosity in the form of dissolution features (sinkholes) (Twidale and Bourne, 2000).

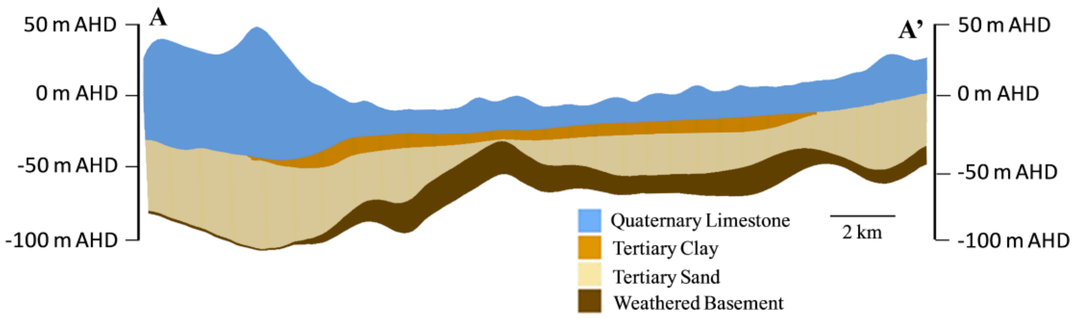

Figure 2.4 – Cross section stratigraphy through A-A’ showing the main geologic strata of the USB (modified from Werner et al., 2011). The cross-section alignment is shown in Figure 2.1, with a general direction SW-NE

Groundwater abstraction from USB is undertaken solely by the water utility organisation SA Water Corporation, who extracts roughly $7x10^6$ m^3/year of high quality groundwater from the Quaternary sediments (Werner et al., 2011). The hydraulic conductivity of the Quaternary sequence reaches up to 2000 m/d locally, reflecting karst aquifer characteristics (Sibenaler, 1976). The high permeability of the aquifer produces relatively flat water tables, which fluctuate in response to the annual cycle of wet-dry seasons. Water table responses to individual rainfall events are of small magnitude (i.e. in the order of a few millimetres), likely due to the coastal proximity and hydraulic connection to the sea, i.e. the coastal head boundary tends to control water levels and high hydraulic conductivities extend this water-level control significant distances inland.

Based on all the available physiographic data, a generalised conceptual model of the recharge processes in the basin was developed. Sinkholes are believed to be important infiltration pathways at the land surface, but the contribution of sinkhole infiltration to the basin’s total recharge, however, is yet to be understood. The sinkhole depths are presently unknown, and so it is not certain whether they directly recharge the water table or simply conduct water deeper into the unsaturated zone. As such, the influence of evapotranspiration on sinkhole recharge is also unknown. Moreover, the degree to which vegetation assemblages draw from the unsaturated and saturated zones is unclear.

2.3 Methodology

2.3.1 Water-table fluctuation analysis

The WTF method uses rainfall-induced water-table rise and the aquifer's specific yield to estimate recharge q_R [LT^{-1}], calculated as (Healy and Cook, 2002):

$$q_R = S_y \frac{\Delta h}{\Delta t} \tag{1.1}$$

where S_y [-] is the specific yield, h [L] is the water-table height and t is time [T]. Water-table rise (Δh) is taken from the extended hydrograph recession to the water level peak of the recharge event (Healy and Cook, 2002). The WTF method has the advantage over some other methods in that it provides insight into transient recharge trends. The existence of preferential flow paths is not a restriction in its application, although the WTF method is limited by difficulties in accurately determining S_y, and additionally, such factors as the presence of entrapped air, changes in barometric pressure and evapotranspiration can influence water-table fluctuations (Healy and Cook, 2002).

The WTF method was applied using monthly groundwater levels of fifteen observation wells with records of varying continuity, spanning the period 1961 to 2007 (DWLBC, 2010). An example of a well hydrograph used in the WTF calculations is given in Figure 2.2. Recharge was estimated for discrete water-table rise-and-decline events (typically spanning several months) from time series data, and these were then summed to produce annual recharge estimates. Values of S_y for the Quaternary Limestone aquifer are reported by Sibenaler (1976), Zulfic et al. (2007) and in Werner et al. (2011). Sibenaler (1976) estimated S_y from eight pumping tests, obtaining an average of 0.13 (with values ranging from 0.03 to 0.35). Zulfic et al. (2007) used a range of 0.10 to 0.30 as possible input S_y values in a groundwater model of USB's Quaternary Limestone aquifer, and through calibration of the model they established S_y zones of 0.20 and 0.30, giving a basin average of about 0.25. Recently, Werner et al. (2011) calibrated a lumped water balance model of USB and arrived at S_y values ranging from 0.14 to 0.20. A plausible S_y range for the purposes of this study was taken as 0.12 to 0.25.

The WTF method relies on water-table fluctuations, which can reflect phenomena other than groundwater recharge, for example pumping (e.g. Healy and Cook, 2002). As such, time variant pumping rates can impact the application of the WTF approach to estimate recharge (Cuthbert, 2010). For example, water-table rise is expected during periods of decreased pumping, leading to a potential overestimation of recharge (Cuthbert, 2010). Further, accuracy of the method depends on the diffusivity of the aquifer, the location of the borehole in question, and the distribution in time and space of the abstraction wells. USB has been subject to considerable groundwater abstraction since November 1976, and Werner et al. (2011) estimated that pumping was a significant component (26%) of the cumulative outflow from the Quaternary Limestone aquifer during the period of 1967-2007. Modifications of the WTF method to account for specific site characteristics or to meet particular objectives have been proposed before (e.g. Crosbie et al., 2005; Moon et al., 2004; Cuthbert, 2010). Here, a modification of the WTF approach to account for the effect of pumping seasonality on WTF recharge estimations is proposed, where the objective is to evaluate the potential overestimation of recharge arising from neglecting the effect of pumping in USB.

The abstraction of groundwater is well monitored (Werner et al., 2011) and there is a general inverse relationship between pumping and rainfall, whereby pumping decreases during periods of sustained rainfall (and recharge). An overestimation of recharge is expected if Equation (1.1) is applied without accounting for the influence of pumping. A modified version of the WTF method was subsequently adopted in which a correction to account for pumping-induced water-table rise, q_{R*} [LT^{-1}], is calculated as:

$$q_{R*} = \frac{Q_{PRP} - Q_{DPR}}{ASy} \tag{1.2}$$

where Q_{PRP} [L^3T^{-1}] is the average pumping rate during successive months of no recharge (i.e. when groundwater levels are falling), Q_{DRP} [L^3T^{-1}] is the average pumping rate during successive months of recharge (i.e. when groundwater levels are rising) and A [L^2] is the planar area of the basin. The corrected recharge is then given as $q_R - q_{R*}$, if q_R is obtained using Equation (1.1).

2.3.2 Chloride mass balance approach

The CMB method has been widely used to estimate groundwater recharge in semi-arid regions (e.g. Wood and Sanford, 1995; Scanlon et al., 2006), i.e. of similar climate to USB. The method exploits the fact that Cl in precipitation becomes concentrated by evapotranspiration, such that the increase of the Cl concentration in groundwater relative to rainwater is then a measure of the proportion of rainfall that has evaporated. This assumes that: (i) the only source of groundwater Cl is atmospheric deposition (i.e. dry deposition and rainfall Cl combined), (ii) there is no surface runoff from the recharge area, and (iii) the atmospheric Cl deposition is in steady state. The mean annual recharge flux, q_R [LT^{-1}], is calculated by (e.g. Wood and Sanford, 1995):

$$q_R = \frac{PC_{P+D}}{C_{GW}} \tag{1.3}$$

where P [LT^{-1}] is the long-term average rainfall, C_{P+D} [ML^{-3}] is the representative mean Cl concentration in rainwater including contributions from dry deposition, and C_{GW} [ML^{-3}] is the Cl concentration of groundwater in the recharge area. The assumptions required in the application of the CMB method are considered to be satisfied in USB. That is, it is assumed that there are no sinks or sources of Cl in the rock matrix, the atmospheric deposition is the only source of Cl to the system, and rainfall is partitioned into only infiltration and evapotranspiration at the basin scale.

As USB is bordered by the ocean, a strong dependency of C_{P+D} on the distance from the coast is expected. To take this dependency into account, the empirical formula developed by Hutton (1976) was used:

$$C_{P+D} = 35.45\left(\frac{0.99}{d^{0.25}} - 0.23\right) \tag{1.4}$$

where d [L] is the distance from the coastline in km. This relationship was obtained for an area in Southeast Australia with an average annual precipitation of > 500 mm/year, and where the distance from the coast was between 0.5 and 300 km (Hutton, 1976). Cl concentrations calculated using Equation (1.4) differed by no more than 5% from measured values in rainfall samples near ULE 109 (d = 6.1 km, Figure 2.1), obtained during July 1988 to March 1992 (Evans, 1997) and rainfall samples next to ULE 101 (d = 2.3 km, Figure 2.1), obtained during May 2008 to March 2009.

The CMB method can be applied using values for C_{GW} that represent either: (i) averaged Cl concentrations over a number of samples in an aquifer (Wood and Sanford, 1995), or (ii) Cl concentrations from individual wells (Sibanda et al., 2009). The first approach yields a basin-wide average but it cannot be readily applied to USB as the Cl deposition rate varies spatially due to the vicinity of the sea. Therefore, the second approach was adopted. The difficulty with this method is that the calculated recharge value applies to an area located somewhere up-gradient of the observation well, i.e. the sampled groundwater has travelled a certain distance through the aquifer since it was recharged. In an attempt to correct for this effect, the following steps were taken to estimate the position of the recharge area corresponding to an observation well. First, the age of the groundwater sample was determined based on its CFC concentration (see below). Then, using typical values of the hydraulic conductivity (100 to 500 m/d) and porosity (0.15 to 0.3) of the aquifer, and the local hydraulic head gradient, the groundwater flow velocity was estimated based on Darcy's law. The length of the flow path up-gradient of the observation well was found by multiplying the flow velocity by the CFC-derived groundwater age. Finally the up-gradient direction from the sampled well to the recharge area was determined by using groundwater pathways derived from hydraulic head contours. Values of C_{GW} were taken from Evans (1997) (sampled in 1993 and 1994), and from DWLBC (2010) (sampled from 1961 to 2004).

2.3.3 Groundwater dating using CFCs

Groundwater CFC concentrations can be used to calculate the time that has elapsed since a water parcel crossed the water table and thus became isolated from the atmosphere (Plummer and Busenberg, 2000). Atmospheric concentrations of CFC-11, CFC-12 and CFC-113 increased continuously between 1945 and 1990 (Plummer and Busenberg, 2006), so that the moment of recharge can be determined by matching the equilibrium partial pressure of a particular CFC in a groundwater sample with its historical concentration in the atmosphere. The basic assumptions underlying this method are that the historical CFC composition of air in the recharge area is known and that the composition of unsaturated zone air resembles that of the atmosphere. Details of the method are given in Plummer et al. (2006).

The application of CFCs to date groundwater was based on data collected by Evans (1997), who sampled seven wells for CFC-11 and CFC-12 in 1995. The CFC's Southern Hemisphere historical air concentration curve (USGS, 2010) was used in the groundwater dating calculations. Preliminary results showed that groundwater ages obtained using CFC-11 are older relative to groundwater ages dated with CFC-12, which indicates that CFC-11 has been subject to degradation (Cook et al., 1995; Plummer and Busenberg, 2000). For this reason only the apparent ages calculated from CFC-12 were used.

Under idealized conditions, such as a constant thickness aquifer receiving uniform recharge and with homogeneous hydraulic properties, an exponential increase in the groundwater age would, theoretically, be observed with depth below the water table, and the groundwater age can be used to infer the recharge rate (Solomon et al., 2006). Such conditions are thought not to apply in USB, and rather there is little evidence of vertical stratification in water quality parameters. This is likely due to various factors, including the highly heterogeneous nature of the aquifer, and the variability in recharge. The long screen lengths of the majority of monitoring wells would further mask any vertical stratification, if present. The calculated ages were used in approximating groundwater flow lengths and expected locations of upstream recharge areas attributable to the groundwater obtained from wells used in applying the CMB method. That is, simple Darcy's law calculations were undertaken to approximate the groundwater travel distance using estimates of hydraulic conductivity, hydraulic gradient, porosity and the groundwater age.

2.3.4 ^{18}O, Cl and Br data analysis

The Cl and ^{18}O data were used in this study to infer the evapotranspiration processes, and in which way they impact the groundwater recharge. The Cl/Br ratios were used to confirm the origin of the recharging water, and to underpin the use of the CMB method.

The Cl and Br concentrations and $\delta^{18}O$ values were adopted from Evans (1997) (25 groundwater samples collected in 1993 and 1994). Evans (1997) collected five samples of rainwater for determination of $\delta^{18}O$. These were compared with the rainfall isotopic data provided for Adelaide by the International Atomic Energy Agency (IAEA) Global Network of Isotopes in Precipitation (GNIP) service (IAEA and WMO, 2005); only complete annual data sets from the GNIP database were used.

2.4 Results and discussion

2.4.1 Conceptual model of groundwater recharge mechanisms

The available chemical and isotope data provide important insights into the processes that contribute to groundwater recharge. The Cl concentrations in groundwater samples from the Quaternary Limestone aquifer range from 100 to 220 mg/L. No spatial trends in Cl concentrations are apparent, and despite the Cl concentrations in the underlying Tertiary Sand aquifer being generally higher (ranging from 137 to 524 mg/L; data from Evans, 1997), no increase of Cl concentrations with depth within the Quaternary aquifer is observed. It is therefore assumed that any mixing of waters from these two aquifers is insufficient to preclude application of the CMB method to estimate recharge.

Figure 2.5 shows the $\delta^{18}O$ values versus Cl concentrations in groundwater. This plot shows that: (i) there appears to be a minute increase of $\delta^{18}O$ over the observed range of Cl concentrations, and (ii) no groundwater samples exhibit Cl concentrations below 100 mg/L. Ranges in $\delta^{18}O$ values (IAEA and WMO, 2005) and Cl concentrations for precipitation are also indicated in Figure 2.5. The averages of bulk rainfall $\delta^{18}O$ from IAEA and WMO (2005) and from Evans (1997) compare well, being -4.3‰ and -4.2‰ respectively.

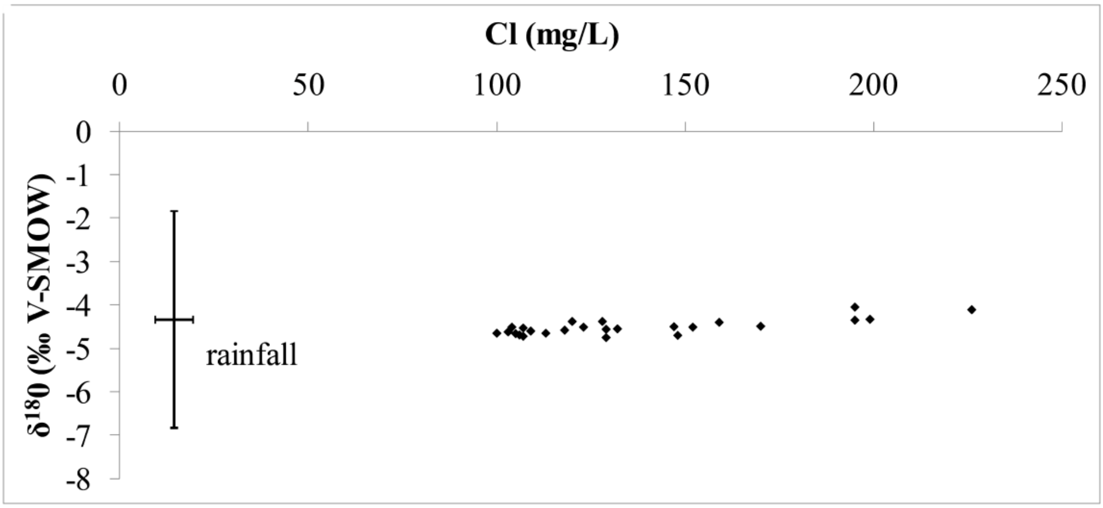

Figure 2.5 – $\delta^{18}O$ (‰ V-SMOW) values versus Cl concentrations of groundwater samples (data from Evans, 1997). Ranges from bulk rainfall samples have also been indicated - the error bars indicate the standard deviation, crossing at the mean values

Cl/Br ratios in both the rainfall and the groundwater samples are all very close to the Cl/Br ratio of seawater of 288 (using concentrations expressed in mg/L), indicating that dissolved salts in rainwater and groundwater are marine derived. This observation is consistent with the vicinity of the ocean, the land use (no application of fertilizers and pesticides) and the lithology of the basin (i.e. absence of Cl-bearing rocks). It rules out the contribution of Cl from sources other than both rainfall and dry deposition of marine aerosols, thereby fulfilling one of the key assumptions for the application of the CMB method.

It should be noted that the Cl concentrations of bulk rainfall integrate both wet and dry deposition, so that the observed Cl concentration gap between groundwater samples and rainwater (Figure 2.5) cannot be attributed to the dry deposition and subsequent dissolution of Cl-bearing particles from the atmosphere. This, and the absence of other sources of dissolved Cl as inferred from the Cl/Br ratio, implies that the increases in groundwater Cl relative to bulk rainfall are due to evapotranspiration. Moreover, the observed lack of significant enrichment of ^{18}O as Cl increases (Figure 2.5), is interpreted to reflect one or a combination of two processes, namely: (i) transpiration dominates over evaporation (plant root water uptake is not accompanied by isotopic fractionation; e.g. Wood and Sanford, 1995; Ingraham et al,. 1998; Geyh, 2001), and (ii) at times, complete evaporation of rainfall occurs and the precipitated salts are washed down and redissolved during wet phases when recharge occurs. The latter process may occur at the land surface during overland flow of rainfall or within the sinkholes themselves after infiltration. What remains unclear is whether the vegetation, the mallee trees in particular, draw their water from the unsaturated zone or whether their roots have access to groundwater. No information exists on rooting depths of these trees in USB.

The absence of intermediate data points between the rainfall and groundwater data points in Figure 2.5 is interpreted to reflect the fact that any rainfall that results in recharge first undergoes either or both of these processes (complete evaporation and transpiration) before reaching the water table (or the screens of the monitoring wells). Most recharge occurs in winter, as can be inferred from the water level observations (Figure 2.2). The observed gap between Cl concentration of rainfall and groundwater should be taken as an indication that the contribution of sinkhole-channelled rainfall that escapes evapotranspiration contributes only little to the total recharge amount, as otherwise more intermediate data points would be observed in Figure 2.5. This does not preclude that recharge rates below individual sinkholes are higher than below the undisturbed calcrete, as suggested by Allison et al. (1985) for the Murray Group marine limestone aquifer (Murray River Basin, South Australia), or that there are some localised occurrences of sinkhole recharge directly to the water table.

2.4.2 Recharge estimates based on the WTF and CMB methods

The mean annual recharge obtained by averaging the results of the WTF analysis of the 15 wells varies between: (i) 66 mm/year and 138 mm/year if the pumping correction is not applied, and (ii) 47 mm/year and 129 mm/year if the pumping correction is applied. Recharge ranges reflect the adopted range in S_y, which imparts a significant uncertainty on WTF-estimated recharge. A further source of uncertainty stems from the potential effects of other components of the basin water balance, including groundwater outflow to the sea and any inflows through the basin boundaries. Although these are in principle considered in the WTF method by extrapolating the hydrograph recession to the groundwater level peak of the recharge event (and using that water level as the minimum in the calculation of Δh; as per Healy and Cook, 2002), this is a rather crude approach that may not accurately take into account the response of the groundwater system to other stresses. While some attempt has been made to modify the WTF approach to account for pumping seasonality, only a rough approximation of the correction is obtained from the simple representation of the pumping effects. Nonetheless, it is clear that the pumping impact is significant and cannot be ignored in the WTF methodology.

Figure 2.6 and Figure 2.7 show the relationship between annual rainfall and annual recharge calculated using the WTF method. Negative recharge values are due to the pumping correction factor applied to the lower S_y case, which for some periods is higher than recharge; this occurred in the years of 1991, 2005 and 2007. Despite there being significant scatter in the data points, correlations are apparent, as expected. Figure 2.6 shows that recharge and rainfall correlate well in some years (e.g. 1964, 1968, 1971, 1977, 1978, 1992), while poor correlation occurs in other years (e.g. 1966, 1972 to 1976, 1983, 1984, 1988, 1995, 1996). Figure 2.7 demonstrates that the same annual rainfall total can produce very different recharge rates (e.g. a rainfall rate of ~650 mm/year is associated with recharge rates of 40 to 112 mm/year, using S_y = 0.12) or that the same recharge rate can be produced by different rainfall rates (e.g. a recharge rate of ~40 mm/year can be produced by rainfall rates of 420 to 750 mm/year). This indicates that the annual rainfall total is not the only factor controlling recharge. Other factors expected to play an important role in the temporal recharge trends include temporal and spatial variability in evapotranspiration, soil water content, and rainfall pattern (i.e. rainfall intensity, duration and location). This is commensurate with the indication from chemical and isotope data that either low-intensity, short-lived rainfall events do not generate recharge due to complete evaporation and/or that the vegetation consumes a significant proportion of the infiltrated rainwater.

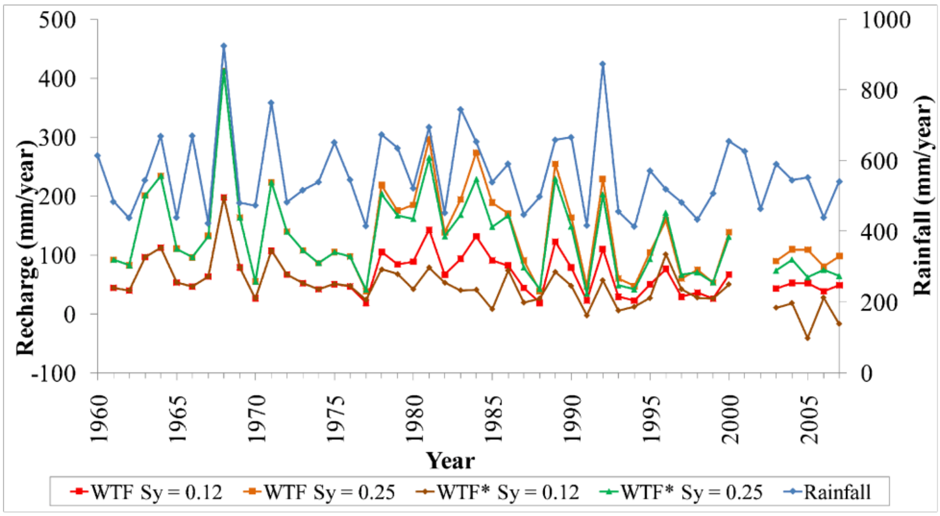

Figure 2.6 – Temporal variability of rainfall and recharge using the standard WTF approach and the approach taking into account the pumping effect in the WTF analysis, symbolized by WTF*. Insufficient data are available for 2001 and 2002

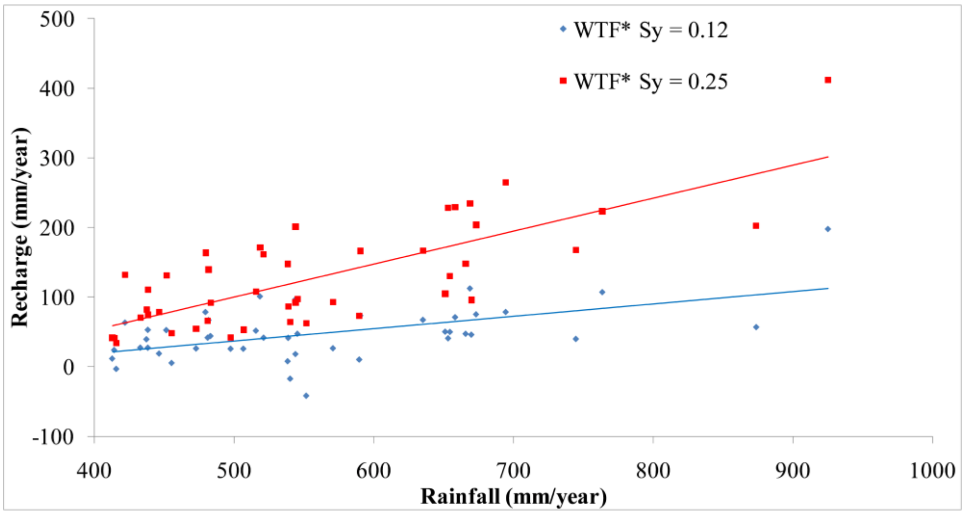

Figure 2.7 – Relationship between annual recharge using the modified WTF approach (WTF*), and rainfall in USB

The spatial variations in time-averaged recharge values from the CMB approach (modified to account for the location of recharge and the distribution of atmospheric Cl deposition) are given in Figure 2.8. There is a general tendency of higher recharge inferred from wells in the central and southern part of the basin. This general tendency is also observed in the recharge results from the WTF method, although it should be kept in mind that these results rely on the assumption of a spatially-constant value of S_y. Regions of calcrete and limited vegetation (see Figure 2.3) are by-and-large situated north-east of wells showing higher recharge rates.

Uncertainties in CMB recharge estimates are linearly dependent on Cl deposition (Scanlon et al., 2006). To account for this uncertainty, a range of possible chloride deposition values was tested in the CMB calculations. CFC-based groundwater ages from seven bores averaged 11 to 14 years, ranging from 9 to 17 years (depending on recharge temperature), equating to infiltrating dates ranging from 1978 to 1986 considering that CFC measurements were taken in 1995 (Evans, 1997). This translates to minimum and maximum distance offsets between wells and recharge areas equal to 0.9 km to 17.3 km, averaging 1.1 km to 13.9 km (depending on the groundwater ages and hydraulic properties adopted in the calculations). The CMB method was applied to these 7 wells using Cl atmospheric deposition rates: (i) at each well location, (ii) at the minimum offset distance upstream of each well, and (iii) at the upstream aquifer boundary (i.e. the maximum offset distances cross the upstream aquifer boundary). The basin-averaged CMB recharge estimates taking atmospheric Cl deposition at the well location is 68 mm/year. If consideration is given to the upstream distance-offset in Cl deposition, CMB recharge estimates reduce by 9 to 32% (accounting for spatial variabilities in parameters and associated uncertainties). These percentages were then applied to all the wells (including those with no CFC data), allowing for testing of the influence of spatial Cl distribution in the CMB recharge estimates across the basin. This produced best-estimate values of basin-averaged recharge of 52 to 63 mm/year, which falls towards the lower end of the pumping-corrected WTF range of 47 to 129 mm/year. Figure 2.8 illustrates distributions of CMB recharge estimates for the lower (52 mm/year) and upper (63 mm/year) bounds.

Contrary to the findings of Crosbie et al. (2010), for some 172 recharge studies across Australia, that estimates based on the WTF method were in general much greater than those based on the CMB method; in this particular case it was found that both approaches produce overlapping recharge estimates. Crosbie et al. (2010) justify the WTF-CMB discrepancy to timeframe differences and to the potential contribution of transpiration from the saturated zone (accounted for in the CMB method, but not in the WTF approach). In this study, both methods were applied in a consistent timeframe (i.e. groundwater in USB is young (11 to 14 years old) and both Cl concentrations and groundwater fluctuations were analysed for the period 1960's to 2000's). Furthermore, USB is influenced by factors which modify the water table behaviour (and therefore WTF estimates of recharge) such as pumping and groundwater discharge to the sea, and these may impose stronger controls than transpiration effects, given the USB specific conditions. Also, the uncertainty in S_y is clearly a major factor in applying the WTF method that potentially masks some of these other uncertain impacts.

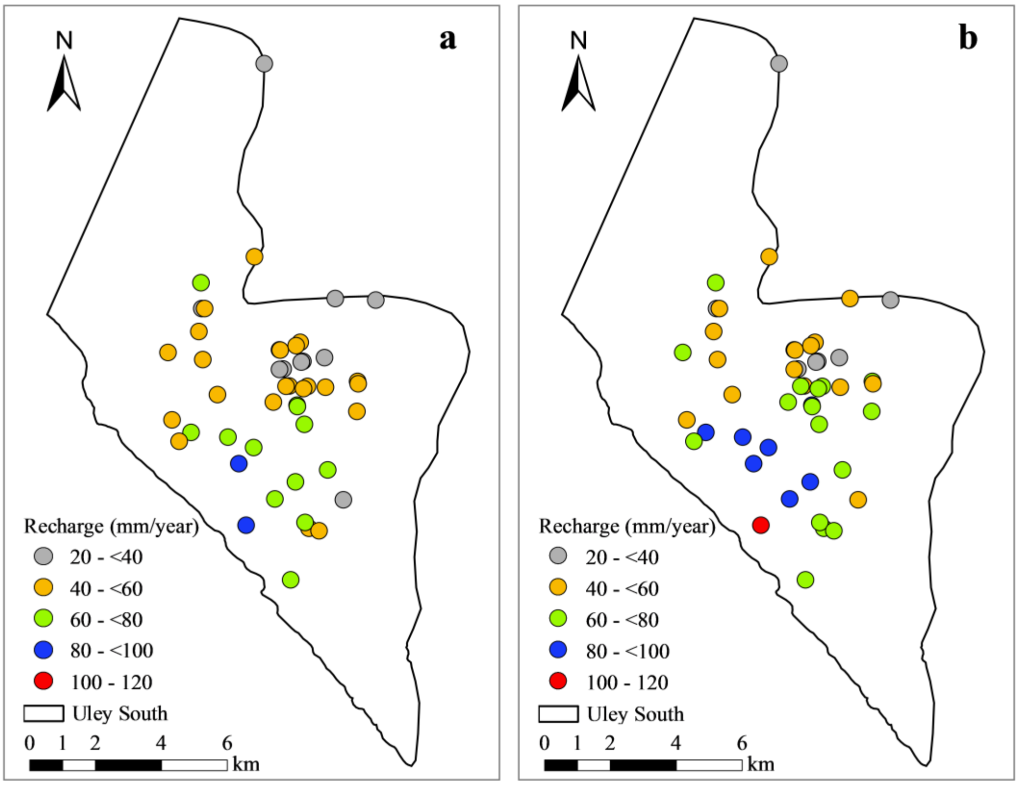

Figure 2.8 – Spatial distribution of CMB recharge averaging (a) 52 mm/year (lower bound estimate) and (b) 63 mm/year (upper bound estimate)

There are several additional sources of uncertainty that may bias the CMB method as applied to the USB. For example, Cl deposition rates (the term PC_{P+D} in Equation (1.3)) based on Cl concentrations measured in bulk precipitation from open (un-vegetated) sites are known to underestimate Cl deposition rates under vegetated sites (Beier and Gundersen, 1989; Moreno et al., 2001; Neary and Gizyn, 1994; Knulst, 2004), thereby underestimating CMB recharge. The magnitude of this effect is dependent on the climate, vegetation type, the extent of the vegetated area, and distance to the coast. For example, in coastal dunes in Holland covered by shrubs, the effect was found to be only 10% (Stuyfzand, 1993). Casartelli et al. (2008) found that the increase in Cl deposition in subtropical forested areas with *Eucalyptus spp.* was 50% in a site away from the coast and over 100% in a site close to the coast. On the other hand, vegetation will reduce the rate of rainfall reaching the land surface through canopy interception, and enhanced transpiration is expected in vegetated areas, and these both serve to decrease the CMB recharge estimates (e.g. by reducing P in Equation (1.3)). The majority of the wells and/or respective recharge areas are located in lightly or un-vegetated sites (i.e. the distribution of monitoring does not reflect the basin's distribution in soil-vegetation), thereby not being influenced by the vegetation effect in Cl atmospheric deposition. Further research is warranted to assess the extent to which vegetation causes CMB-estimates of recharge to be biased. Hypothetically, if a 50% influence of vegetation in C_{P+D} occurs over 57% of the basin (i.e. under highly vegetated areas; see Figure 2.3b), this would result in CMB estimates being higher by 28%.

The seasonality in the groundwater salinity of production bores was also considered as a potential source of error in applying the CMB approach - the salinity of abstracted groundwater is between 8 and 15% higher on average in summer than in winter. That is, recharge estimates based on Cl concentrations of groundwater samples taking during the summer may be lower than those taken during winter months, given C_{GW} variations between the seasons. Also, C_{GW} in Equation (1.3) strictly represents the Cl concentration at the water table. The temporal and spatial variability in recharge and Cl deposition rate is expected to create some amount of vertical stratification in groundwater Cl concentrations, albeit vertical variations in groundwater salinity within the Quaternary Limestone were not apparent in the available water chemistry data. To test the likely scale of temporal groundwater salinity effects, the CMB method was applied by separately considering samples collected at different depths below the water table (i.e. < 3 m, < 10 m, and all the samples). The effect of the sampling depth appears to be negligible (< 3%) in the recharge estimates using the CMB method.

2.5 Conclusions

In this study, a conceptual model of groundwater recharge to a heterogeneous coastal aquifer subject to pumping, seasonal rainfall and evaporation stresses was developed. A plausible description of recharge processes arising from the available data is given as: (i) Cl in the system is essentially from atmospheric origins and influenced by proximity to the ocean; (ii) a substantial proportion of rainfall occurring in dryer months is probably subjected to complete evaporation, with precipitated salts carried into the aquifer during wetter months; (iii) the primary function of sinkholes at the basin scale is to redistribute water into the unsaturated zone rather than cause direct recharge to the aquifer; (iv) soil water originating from (iii) is subject to transpiration more so than evaporation.

The WTF and CMB methods applied in this paper are modified forms of previous applications. Modification to the WTF method involved a simple correction for water-table rise caused by pumping reductions, producing a basin-averaged and time-averaged (1967-2007) WTF recharge estimate that was lower by up to 30% than the “traditional” WTF method. The CMB approach was adapted to consider the location of the recharge relative to the position of the well, based on CFC ages and groundwater flow distances and directions, resulting in a basin-averaged recharge 9% to 32% lower than if the Cl deposition was considered at the well location. Although the accuracy of the alternative recharge estimations cannot be validated, it was demonstrated the importance of considering local site conditions in applying these common approaches to recharge estimation. The results indicate that, potentially for this basin, an overestimate in WTF-based recharge may arise from neglecting pumping effects, and over- or underestimates in CMB-based recharge may occur if Cl deposition factors are ignored.

Estimates of the long-term and spatially averaged recharge to the Uley South Quaternary Limestone were: (i) 52 to 63 mm/year from the CMB approach, and (ii) 47 to 129 mm/year from the WTF approach. Ranges reflect uncertainty in input parameters, although the assumptions regarding vegetation effects and salinity seasonality were not considered as they were not able to be quantified. The observed decrease in the water levels, especially in areas of shallow water tables, may have altered the groundwater recharge pattern due to associated changes in evapotranspiration. This effect requires additional investigation. The range in possible WTF recharge values is a function of the uncertainty in S_y. Although this method provides a large range of possible recharge values (i.e. larger than the CMB method), it has the advantage of providing information about the time variability of recharge, which cannot be obtained with the CMB method. The range of recharge results presented here is smaller than previous recharge estimations for the basin (i.e. 40 to 200 mm/year), which is indicative of the contribution from extending the recharge conceptual model beyond pre-existing versions. Furthermore, the current recharge value of 140 mm/year adopted in setting groundwater management policies for the basin is likely an overestimate of recharge. The uncertainty in recharge estimates could be reduced through further work, aimed at providing a better understanding of the influence of vegetation on atmospheric Cl deposition, a better characterization of Cl concentration in groundwater across the basin and an improved characterisation of specific yield, which would collectively provide more accurate and reliable data for the application of the CMB and WTF approaches. Likewise, a more detailed characterization of spatial and temporal variations of hydrochemistry and hydraulic parameters, both in saturated and in unsaturated zones, would enhance the understanding of the processes affecting recharge in USB.

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3 Influence of model conceptualisation on one-dimensional recharge quantification - Uley South, South Australia

Abstract

Model conceptualisation is a key source of uncertainty in one-dimensional recharge modelling. In this study, the effect of different conceptualisations on transient recharge predictions for the semi-arid Uley South Basin, South Australia, were investigated. One-dimensional unsaturated zone modelling was used to quantify the effect of variations of (i) lithological complexity of the unsaturated zone, and (ii) representation of preferential flow pathways. The simulations considered ranges of water table depths, vegetation characteristics, and top soil thicknesses representative for the study area. Complex lithological profiles were more sensitive to the selected vegetation characteristics and water table depth. Scenarios considering runoff infiltration into, and preferential flow through sinkholes resulted in higher and faster recharge rates. A comparison of modelled and field-based recharge estimates indicated that: (i) the model simulated plausible recharge rates, (ii) only the models with preferential flow correctly reproduced the timing of recharge, and (iii) preferential flow is probably redistributed in the unsaturated zone rather than passing to the water table directly. Because different but equally plausible conceptual models produce widely varying recharge rates, field-based recharge estimates are essential to constrain the modelling results.

3.1 Introduction

A reasonable approximation of spatial and temporal distributions of groundwater recharge at the basin scale is a necessary precursor to effective groundwater management. No single method adequately characterises recharge (e.g. Flint et al., 2002; Scanlon et al., 2002a; Healy, 2010), which often makes choosing recharge estimation techniques difficult (Scanlon et al., 2002a). The assumptions required to develop conceptual and mathematical models of hydrological processes can have important consequences for predicting recharge rates. Although such practices are not common, the use of multiple conceptual models is recommended as means to assess uncertainties in modelling (Bredehoeft, 2005; Seifert et al., 2008).

Keese et al. (2005) advocate the use of one-dimensional modelling to calculate recharge, as it is the only approach capable of predicting future recharge patterns, and it enables examination of the relative importance of various controlling factors. They highlight that the primary difficulties in recharge modelling are data availability and the selection of model parameters, especially at regional scales. Similarly, Scanlon et al. (2002a) suggest that the range of recharge rates obtained using numerical modelling can vary greatly due to difficulties in characterising hydraulic conductivity and nonlinearities between hydraulic conductivity, water content, and matric potential.

Previous studies have attempted to address model complexities by exploring the sensitivities of recharge predictions to changes in model inputs, thereby providing insight into model uncertainty, as well as guidance on data collection (e.g. Scanlon et al., 2003). For example, Keese et al. (2005) studied the factors that control recharge at the scale of the state of Texas (USA). A sensitivity analysis was undertaken to examine the effect of variations of vegetation, unsaturated zone texture, climatic inputs and profile depth on predicted recharge. Their models were particularly sensitive to vegetation characteristics, potential evapotranspiration rates and the unsaturated zone texture. Smerdon et al. (2008) also explored the sensitivity of one-dimensional recharge models for shallow glacial outwash aquifers in Canada. They concluded that groundwater recharge at the study scale (about 5 km^2) was mainly influenced by climate and the depth to water table (DWT). Scanlon et al. (2010) found that the vegetation rooting depth (RD) was a sensitive parameter in obtaining simulated recharge values that were consistent with field estimates in the Thar Desert in India.

From these studies it follows that the most sensitive parameters and processes in recharge models depend on the physical characteristics of the system, which are always uncertain. Moreover, complex processes such as preferential flow, which exert a strong control on recharge (e.g. Ireson and Butler, 2011), are often not simulated in regional scale studies (e.g. Keese et al., 2005). Therefore, model testing needs to include variations to the conceptual model to investigate the manner in which complex recharge processes are represented (Healy, 2010; Seifert et al., 2008).

The recharge processes in the semi-arid Uley South Basin (USB), South Australia, and the recharge rates based on the chloride mass-balance (CMB) and water-table fluctuation (WTF) methods, have been investigated previously by Ordens et al. (2012). A key knowledge gap identified in the analysis of Ordens et al. (2012) was the fate of surface runoff and its infiltration via sinkholes. There is a need to develop a predictive model of recharge to USB for management purposes (Werner et al., 2011) with the ability to estimate recharge in areas and at times that are not presently estimable from field data. Data paucity prevents exact field conditions to be replicated in models, and rather, different scenarios corresponding to plausible conceptualisations of the recharge processes, thought to occur at various locations across the basin, were constructed.

The objective of the present study is to investigate the influence of variants of the conceptualisation of recharge processes in USB on recharge predictions based on one-dimensional unsaturated flow modelling. A novel aspect is the analysis of the effect of different conceptual models on the recharge quantification. The focus was on two aspects of model conceptualisation: (i) the complexity of the unsaturated zone lithology (simple versus complex profiles of the unsaturated zone lithology); and (ii) runoff infiltration and preferential flow (matrix flow versus bypass flow). Due to the hard calcareous nature of the terrain, there is a difficulty in obtaining unsaturated zone data that can be used to constrain and/or validate one-dimensional models. Therefore the model outcomes were tested against the recharge estimates of Ordens et al. (2012) that were based on the WTF method.

3.2 Study Area

3.2.1 Physical setting

The physical setting of USB has been described by Harrington et al. (2006), Werner et al. (2011) and Ordens et al. (2012). Here, only a succinct description of the setting is provided. USB is a topographically closed surface drainage basin of about 129 km^2, with a flat to undulating centre at an elevation of less than 10 m AHD (Australian Height Datum, approximately equal to mean sea level) (Figure 3.1). Abstraction of groundwater from the basin supplies around 70% of the water demand of the Eyre Peninsula's population of around 33,500 (Zulfic et al., 2007); Werner et al. (2011) reports a groundwater abstraction from USB of around 175 GL for the period 1967–2007. Groundwater is also discharged to the ocean (Werner et al., 2011). The subsurface of the basin can be divided into three hydrogeological units (Harrington et al., 2006): (i) an unconfined Quaternary Limestone (QL) aquifer; (ii) a Tertiary clay aquitard; and (iii) a confined to unconfined Tertiary sand aquifer. The QL extends across the entire basin, and therefore the recharge to this aquifer is the subject of the current study. The QL aquifer consists of a heterogeneous combination of unconsolidated to poorly consolidated aeolian sediments and limestone of varying hardness and porosity.

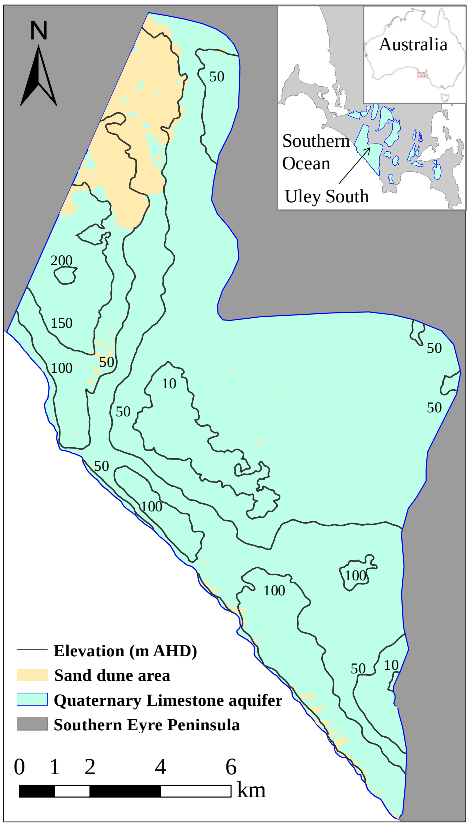

Figure 3.1 – Map showing the extent of USB, surface elevations and locations of the sand dune areas

Secondary cementation, occurring as calcrete, as well as secondary porosity in the form of dissolution features is extensive across the QL (Twidale and Bourne, 2000). In addition to those dissolution features, thousands of sinkholes exist across the basin in calcrete layers, and their diameter varies from a few centimetres to several decimetres. A thin layer of calcareous or loamy soil overlays the QL, although it is absent in places (Harrington et al., 2006). Thicker soils, however, can be found under vegetated sites, reaching tens of centimetres. Here, 'soil' is meant to denote the top part of the unsaturated zone where the presence of organic matter is evident. Part of the QL is covered by calcareous sand dunes in which no soils have developed and no vegetation is present (Figure 3.1).

The vegetation is variable across the basin, and has been described by Swaffer et al. (2013). Mallee trees (*Eucalyptus diversifolia* and *Eucalyptus gracilis*) occur as dense stands above 30 m AHD, and as isolated stands below this altitude. She-oak trees (*Allocasuarina verticillata*) occur as isolated stands, and extensive areas contain fallen, deceased trees from this species. Portions of the basin are covered with shrub (e.g. *Leucopogon parviflorus* and *Melaleuca lanceolata*).

Climate data for the study area were obtained from the Big Swamp monitoring station (18017) (DSITIA, 2013), located 18 km NE of the basin, where daily data, either measured or interpolated, are available dating back to 1889. The climate is semi-arid, characterized by cool wet winters and hot dry summers (Zulfic et al., 2007). The average annual rainfall is 560 mm/year, the average modelled FAO56 reference evapotranspiration (Allen et al., 1998) is 1084 mm/year, and the average modelled pan evaporation (e.g. Jones, 1992) is 1547 mm/year (DSITIA, 2013).

3.2.2 Recharge processes

The recharge processes were investigated by Ordens et al. (2012) using hydrochemical, isotopic and water level data. At the basin scale, rainfall within USB is partitioned into evapotranspiration and groundwater recharge only, as there is no surface water drainage to the sea. Surface runoff within the basin is highly ephemeral, occurring only after prolonged rainfall events and persisting for up to a few hundred metres before infiltrating through sinkholes and other dissolution features. Ordens et al. (2012) offered three hypotheses for the main recharge processes occurring in the basin that are likely to co-exist: (i) a substantial proportion of rainfall occurring in dryer months may be subjected to complete evaporation; (ii) soil moisture and groundwater are subject to transpiration more so than to evaporation; and (iii) dissolution features do not constitute a direct pathway from the surface to the water table, but are likely to distribute water deeper in the unsaturated zone that can be then accessed by the vegetation and subjected to transpiration, although some direct recharge cannot be totally excluded.

Based on the CMB, Ordens et al. (2012) estimated temporally and spatially averaged recharge rates to range from 52 to 68 mm/year. They also determined that CMB time-averaged recharge rates using chloride data from single wells are spatially variable in the basin, and can be as low as 35 mm/year and as high as 114 mm/year. The authors also found indications that recharge is higher in areas of limited vegetation, as expected. Application of the WTF method showed that annual recharge generally follows the rainfall trend (Ordens et al., 2012). The WTF method indicated that, for an average year, recharge occurs predominantly during the winter months of July–September, while the rainfall peak occurs in July.

3.3 Methodology

3.3.1 Model description

The LEACHM code (Hutson, 2005) was used to simulate unsaturated flow. LEACHM uses a finite-difference approach to solve a one-dimensional form of Richards' equation for unsaturated matrix flow:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left(K(\theta)\frac{\partial H}{\partial z} \right) - S \qquad (3.1)$$

where θ is soil moisture content [L^3L^{-3}], K is the hydraulic conductivity [LT^{-1}], H is hydraulic head [L], t is time [T], z is depth [L] (positive downwards), and S represents a source/sink term [T^{-1}]. A modified form of Campbell's (1974) K-θ-h

functions (where *h* is pressure potential), proposed by Hutson and Cass (1987) was used to define unsaturated zone hydraulic relationships.

Surface runoff is treated in LEACHM as follows. Rainfall is partitioned into runoff and potential infiltration according to a slope-based curve number approach (Williams, 1991). The curve number approach is based on empirical relationships that take into account the hydrologic characteristics of the surface and antecedent near-surface soil water content (USDA, 1986). During infiltration, additional runoff can be generated owing to: (i) saturation-excess runoff, whereby the available storage in the soil is exceeded; and (ii) infiltration-excess runoff, whereby rainfall exceeds the infiltration capacity of the soil. The LEACHM code was modified to enable surface runoff water to be transmitted directly into the unsaturated zone at any user-specified depth, which accounts for the rapid bypass flow of surface runoff into the unsaturated zone via sinkholes, or other preferential flow paths. This is achieved through the *S* term in Equation 3.1. If the water content at any node reaches saturation, runoff water is assigned to the segment immediately above.

Unsaturated flow in USB was simulated for the period January 1957 to December 2007. The model cycled through this period twice with the first cycle merely used to generate realistic initial conditions for the second cycle. The boundary condition at the land surface accounted for rainfall and reference evapotranspiration derived from historic daily weather data from Big Swamp weather station. In LEACHM, reference evapotranspiration is partitioned into potential transpiration and potential evaporation depending on the foliage projection cover (FPC) which ranges from 0 (no transpiration) to 1 (no evaporation). Specht and Morgan (1981) define FPC as the proportion of the ground area with foliage (or photosynthetic tissue) vertically above it. Vegetation is parameterised in terms of FPC, a RD distribution and the duration of growth (i.e. periods of active transpiration). Transpiration from soil segments in the root zone is weighted by matric potential and *K*, and a root water potential is iterated until the total root water uptake equals potential transpiration, or until lower limits on root water potential and/or matric potential are reached (Hutson, 2005). Rainfall is apportioned into canopy interception and throughfall according to FPC and an interception capacity. The lower boundary condition was a constant positive pressure potential, adjusted to reflect a desired DWT. Many one-dimensional recharge models use a unit water-flux gradient lower boundary condition (e.g. Scanlon et al., 2002b; Keese et al., 2005). Because DWT is variable across USB, and it is believed to impact the unsaturated zone processes through for example, capillary rise, the fixed-depth water table (e.g. Smerdon et al., 2008) was used here in preference to the unit-gradient boundary condition. Test simulations using a unit-gradient lower boundary condition showed virtually the same results as those using a deep fixed water table.

3.3.2 Model parameters

Outcrops in coastal cliffs (Figure 3.2) and drilling of shallow boreholes (up to a depth of 5 m) revealed that USB's upper subsurface consists of a sequence of alternating calcrete, consolidated to poorly consolidated aeolian calcareous sand, and clay layers. The depths and thicknesses of these layers are variable. Previous understanding of the shallow subsurface lithology based on drillhole data from DEWNR (2012) considered a simple lithological structure of a thin soil layer covering a calcrete layer, underlain by consolidated to poorly consolidated sand. Five unsaturated profiles were therefore simulated to represent variations of lithology at different degrees of complexity. The adopted lithological structures of the different unsaturated zone profiles are represented in Figure 3.3, and are: (i) profile 1 (D0) – sand, homogeneous; (ii) profile 2 (C10) – a 10 cm top soil layer, and a sequence of 1 m calcrete layer, 1 m sand layer, 1 m clay layer, repeated to the bottom of the profile; (iii) profile 3 (C30) – a 30 cm top soil layer, and a sequence of 1 m calcrete layer, 1 m sand layer, 1 m clay layer, repeated to the bottom of the profile; (iv) profile 4 (S10) – a 10 cm top soil layer, a 1 m calcrete layer, and sand to the bottom of the profile; and (v) profile 5 (S30) – a 30 cm top soil layer, a 1 m calcrete layer, and sand to the bottom of the profile. D0 represents the sand dune area (covering ~8% of the basin; Figure 3.1), while C10 to S30 exemplify conditions in the remaining area of the basin. C10 and C30 represent profiles of complex lithological structure, while S10 and S30 represent profiles of simple lithological structure. The different top soil thicknesses (i.e. 10 cm in C10 and S10, 30 cm in C30 and S30) are intended to capture the range of top soil thickness across USB. Owing to the thicker top soil layer in C30 and S30 than in C10 and S10, the underlying layers in C30 and S30 are located 20 cm lower than in C10 and S10. While this difference is subtle it is important to highlight here, as it was found to be sensitive to the calculated recharge in selected variants of simulations of C10 and C30.

Figure 3.2 – Coastal cliff face, of about 100 m, showing classic QL stratigraphy with dune sandsheets separated by calcrete and clayey sediments associated with fossil soil horizons. Resistant ledge-forming units are calcrete horizons associated with fossil soil horizons (Source: Eyre Peninsula Natural Resources Management Board, 2010)

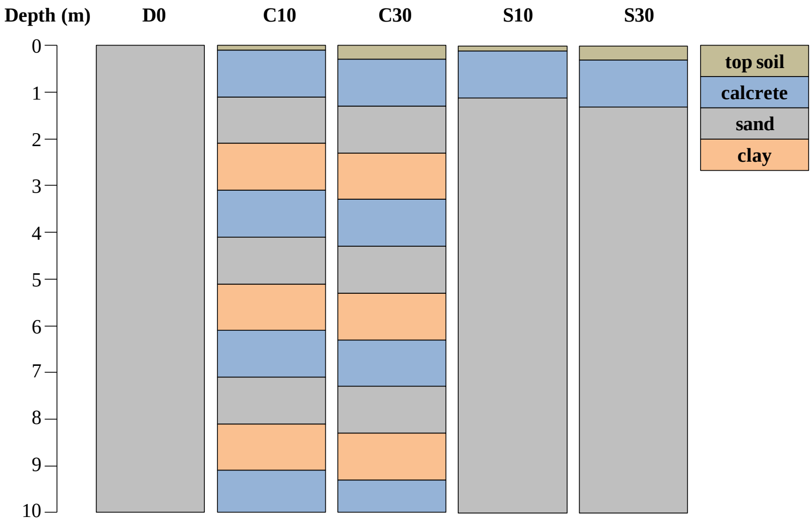

Figure 3.3 – Lithology of the unsaturated zone profiles considered in the model

The profiles were defined to a depth of 10 m, using 100 nodes of 0.1 m each. The various layers in each profile were characterised by Hutson and Cass (1987) *a* and *b* retention function parameters, bulk density, and *K*. The values for the retention function parameters were based on water retention curves measured on disturbed calcrete samples collected during a drilling programme in USB (Table 1). Water retention functions for the remaining lithological types were estimated using typical particle size distributions for loam (top soil), sand and clay soils in conjunction with pedotransfer functions (Hutson, 2005). Bulk density values, measured on sediment clods or calcrete fragments were used to calculate porosity. Starting values of saturated K (K_{sat}) (Table 1) were obtained from Carsel and Parrish (1988) for clay and sand, and from Frances (2008) for calcrete. The top soil K_{sat} values were chosen from the upper range of K_{sat} values proposed by McKenzie and Jacquier (1997) for Australian loam soils. A sensitivity analysis to K_{sat} values was conducted for a number of simulations, and very similar results were found. For example, a 50% change in K_{sat} produced on average a change of 1.6% on long-term recharge rates.

Table 3.1 – Hydraulic parameters for each lithology type simulated in the model

Lithology type	**Campbell parameters**		**Bulk density**	K_{sat}
	a	***b***	**(kg/dm^3)**	**(mm/day)**
Top soil	-0.43	5.05	1.4	2.1 x10^4
Calcrete	-0.82	2.00	2.0	6.4 x 10^2
Sand	-0.52	0.72	1.5	1.0 x 10^4
Clay	-3.00	8.00	1.8	4.8 x 10^1

FPC was measured at one site containing mallee trees, one site containing she-oak trees and two sites containing shrubs using the point quadrat technique (e.g. Specht and Morgan, 1981), and the results indicate an average FPC of 0.5 for mallee trees, 0.3 for she-oak trees and 0.2 for shrubs. From the species analysed in USB, we have only found published values for *Eucalyptus diversifolia* (mallee tree) of FPC = 0.42 (Specht, 1983). Swaffer et al. (2013) measured leaf area indexes across the basin and found values to have small variance. All vegetation types were considered to be active all year around.

No field-based information exists on RD in USB. Although the literature on RD (e.g. Canadell et al., 1996) does not specifically refer the species present in USB, it is known that mallee trees have deep and pervasive root systems (Wright and Ladiges, 1997), while shallower root profiles have been reported for the she-oaks (Sun and Dickinson, 1995). Several RD were simulated for the trees: 3 m, 4 m and 5 m for mallee; and 0.5 m, 1 m and 2 m for she-oak. The shrubs were assumed to have shallower RD, and a depth of 0.3 m was adopted. RD deeper or equal to DWT were not simulated, which consequently resulted in different number of simulations for different DWT and for different RD. Un-vegetated conditions were also simulated. The sand dunes, here represented by D0, are not vegetated, therefore only un-vegetated conditions were simulated for D0, and consequently the results are expected to be significantly different from other profiles.

The weather data available for the model input (Big Swamp weather station) included reference evapotranspiration values calculated using the FAO56 Penman-Monteith equation (Allen et al., 1998), as well as evaporation pan data (e.g. Jones, 1992). Swaffer et al. (2013) installed a weather station in USB and calculated FAO56 reference evapotranspiration for the basin. They correlated those values with values from Big Swamp weather station, and obtained a long-term reference evapotranspiration rate for USB of 1524 mm/year. Although these results are based on short-period data, which constitutes a limitation to their analysis, they indicate that reference evapotranspiration in USB is substantially higher than the FAO56 rates from Big Swamp weather station, but similar to the rates obtained for the evaporation pan for the same weather station. For this reason, Big Swamp's evaporation pan data were used in preference to the lower FAO56 data.

Water-table fluctuations in USB are of small magnitude in response to individual rain events, but show larger seasonal fluctuations (Ordens et al., 2012). DWT is also spatially variable, ranging from shallow water tables of a depth of 2 m, to deeper water tables of depths of tens of meters. A disadvantage of a fixed DWT in one-dimensional modelling is that the water table will not respond dynamically to changes in recharge/discharge. For this reason, a range of DWT was used in the LEACHM simulations to encompass the typical values found in DWT: 2.0 m, 2.5 m, 3.0 m, 3.5 m, 4.0 m, 4.5 m, 5.0 m, 5.5 m, 6.0 m, 7.0 m and 10.0 m. The shallowest water tables are less common in the basin and only occur in limited locations (for example, a DWT of 5 m or shallower only occurs in 17% of the basin). Water tables deeper than 10 m exist in the basin in areas of high surface elevation, but test simulations considering deeper water tables resulted in similar results to the DWT of 10 m.

Slopes are less than 10% in over 90% of the basin. A slope of 10% was used to assess the maximum effect of runoff on recharge. Test simulations adopting an intermediate slope of 5% showed very similar results to the simulations considering 10%. A curve number of 90, considered appropriate for semi-arid rangelands (USDA, 1986), was adopted in all simulations where the curve number routine was used.

A first set of simulations examined the effect of the conceptualisation of the unsaturated zone lithology on recharge, in the absence of runoff and bypass flow (i.e. the curve number routine was not used). A second set of simulations examined the effect of different scenarios of runoff infiltration and preferential flow on recharge. To constrain the number of simulations, profile C10 was chosen for the second set of simulations. The complex profiles were chosen in preference of the simple profiles because they are believed to be more representative of field conditions. A limited number of simulations was conducted for C30 as well, but since the findings from these were similar to those of C10, only the results for C10 are shown for reasons of brevity. Four scenarios of surface runoff infiltration were examined: (i) scenario 1 ($C10_0$) – no bypass flow, i.e. all infiltration passed through the matrix, which represents a situation where preferential flow paths are not considered in the recharge process; (ii) scenario 2 ($C10_{1.5}$) – runoff bypassed the upper 1.5 m, flowing immediately to a sandy layer at that depth, which represents shallow preferential flow paths that bypass only the top calcrete layer; (iii) scenario 3 ($C10_{4.5}$) – runoff bypassed the upper 4.5 m, flowing immediately to a sandy layer at that depth, which represents deep preferential flow paths that transmit water deep into the unsaturated zone (water tables of 4.5 m and shallower were excluded in this scenario); and (iv) scenario 4 ($C10_{WT}$) – runoff bypassed the entire unsaturated zone, and was considered to be direct recharge, which represents deep preferential flow paths that connect the surface with the water table.

3.4 Results and Discussion

3.4.1 Scenario 1: Matrix flow only

The simulations for $C10_0$ resulted in a wide range of annual recharge rates (Figure 3.4). Annual recharge rates are grouped in Figure 3.4 according to vegetation types, RD and unsaturated profile lithology. The variability reflects different DWT, meteorological conditions (rainfall and evapotranspiration) and antecedent unsaturated zone conditions (i.e. profile water distribution at the start of each year). Low or negative (net upward flux from the water table) recharge rates occurred if root and water table depths were such that roots had access to water from the saturated zone through capillary rise. Average annual recharge rates obtained for D0 are of 268 mm/year (48% of rainfall), which constitutes the upper bound of recharge. Un-vegetated conditions resulted in recharge rates of 134 mm/year for C10 and S10, and 99 mm/year for C30 and S30 (24% and 18% of rainfall, respectively). These rates are lower than for D0, because in C10–S30, evaporation is higher due to more water retention in both the top soil and the calcrete layers than in the sand. This finding is in line with results obtained by Keese et al. (2005), who found that monolithic sand profiles resulted in higher recharge rates than variably textured profiles. Furthermore, the recharge rates found in our study for un-vegetated conditions are in the range of those found by Keese et al. (2005) for semi-arid regions of Texas of 43–51% of rainfall for monolithic sands, and 12–29% of rainfall for texturally variable soils (these ranges exclude two regions where Keese et al. (2005) believed to have underestimated recharge).

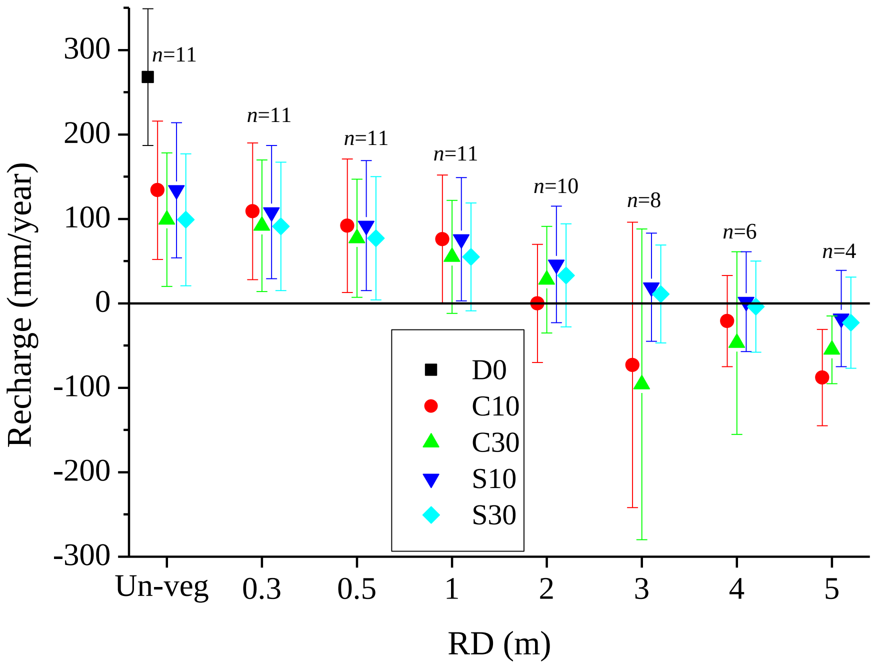

Figure 3.4 – Annual recharge rates for different RD (and vegetation types) and unsaturated zone lithological profiles. The limits of the error bars indicate one standard deviation, and the symbols indicate the average. n is number of simulations (as per Methodology section). The horizontal axis is not to scale

In Figure 3.5, annual recharge rates are grouped according to DWT and unsaturated profile lithology. The variability of annual recharge within each grouping reflects the different vegetation types, RD, the meteorological conditions and antecedent unsaturated zone conditions. The DWT influences annual recharge rates and recharge variability in C10–S30, as expected in one-dimensional models of recharge that adopt a water table as a boundary condition (e.g. Smerdon et al., 2008). This is mainly owing to (i) trees can access groundwater from the capillary zone if the DWT is shallower; and (ii) the recharge process in shallower water table simulations is faster. Nevertheless, DWT are not as important as RD and vegetation types on total recharge (Figures 3.4 and 3.5). D0 recharge values are not influenced by DWT for the range considered (2–10 m) due to the small capillary rise expected in sand and the lack of vegetation, and therefore small penetration of evapotranspiration losses below the land surface. Average annual recharge rates are higher for a DWT of 5 m because for this scenario mallee trees with RD = 5 m were excluded (as per Methodology section).

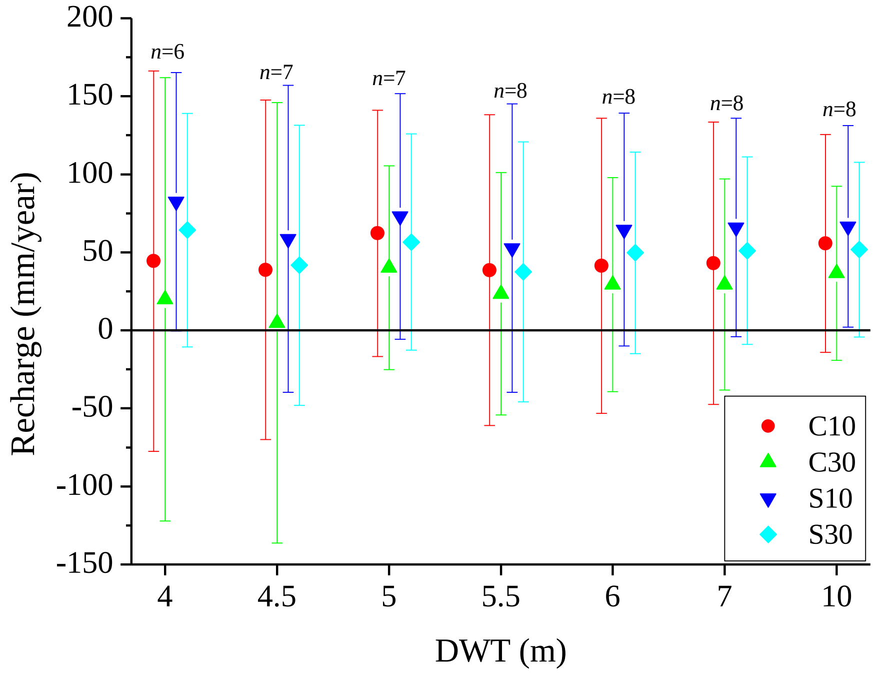

Figure 3.5 – Annual recharge rates for different DWT and unsaturated zone lithological profiles. The limits of the error bars indicate one standard deviation, and the symbols indicate the average. n is number of simulations (as per Methodology section). The horizontal axis is not to scale

Figure 3.5 shows that the simple profiles (S10 and S30) result in higher average annual recharge rates than the complex profiles (C10 and C30); the average of the differences is 80 mm/year. This is a consequence of (i) more water being retained in the clay and calcrete layers of the complex profiles, and being more accessible to vegetation than in the simple profiles; and (ii) stronger capillary rise in calcrete and clay than sand layers, resulting in more water to be drawn up into the root zone via the capillary fringe above the water table.

Figure 3.5 further shows that the profiles of thinner top soils (C10 and S10) result in higher annual recharge rates than the ones with thicker top soils (C30 and S30); the average of the differences is 34 mm/year. This occurs mainly because less soil water is retained (and lost to evapotranspiration) from shallower top soils. Moreover, as explained in the model parameters section above, the difference of 20 cm in the top soil layer thickness causes subtle differences in the relative positions of the capillary fringe and the root zones, which results in different transpiration rates between the simulations. This is borne out in more detail by Figure 3.6, which compares annual recharge rates between C10 and C30, and the results are grouped according to different vegetation types and RD. In the simulations considering mallee trees with RD = 3 m or RD = 4 m, C10 results in higher recharge rates than C30. Conversely, in the simulations considering she-oak trees with RD = 2 m, and mallee trees with RD = 5 m, C30 produces higher recharge rates than C10, in particular when recharge rates are negative.

Negative recharge rates are indicative of high evapotranspiration rates, where roots have access to the groundwater through capillary rise. This has also been found in other modelling studies (e.g. Smerdon et al., 2008; Crosbie et al., 2008). Ordens et al. (2012) concluded that vegetation access of groundwater by vegetation cannot be excluded in USB, but the spatial extent and rates of groundwater transpiration are yet to be understood. Swaffer et al. (2013), based on field data collected during two years in USB, found that the water balance was negative in areas covered by mallee and she-oak trees (i.e. indicating negative recharge), while it was positive in an un-vegetated site (i.e. indicating positive recharge). Nevertheless, Swaffer et al. (2013) were not able to conclude wether the mallee and/or she-oak trees were accessing groundwater or solely unsaturated zone water. Negative recharge typically occurs in those simulations where the combination of DWT and RD leads to the root zone in close proximity to the water table. For deep water tables, recharge can still be zero for certain types of vegetation. This is exemplified by the simulation with DWT = 10 m and mallee trees with RD = 5 m, in which all infiltrating rainfall is captured and lost to evapotranspiration. These modelling results are consistent with findings by Cook et al. (1989), who estimated recharge of less than 0.1 mm/year under mallee vegetation in a semi-arid region in South Australia, based on the unsaturated zone CMB. Similarly, for texturally variable soils, Keese et al. (2005) found modelled recharge to be zero under trees in a semi-arid region. Both Cook et al. (1989) and

Keese et al. (2005) found that where vegetation was absent or sparse, recharge rates were 10–20% of rainfall.

These results are consistent in a qualitative sense with findings by Ordens et al. (2012), who demonstrated that higher groundwater recharge rates are correlated with areas of limited vegetation and that transpiration likely dominates over evaporation from the deeper unsaturated zone and from the saturated zone. The modelling results presented here support those findings, as evaporation only affects the upper part of the profile, and transpiration plays an important role in water uptake. For example, in C10 simulations of DWT = 10 m, transpiration is 30% of evapotranspiration for shrubs, 44% of evapotranspiration for she-oak trees with RD = 2 m and 56% of evapotranspiration for mallee trees with RD = 5 m; in these cases, average annual evapotranspiration increases from 441 mm/year to 528 mm/year and to 550 mm/year, respectively.

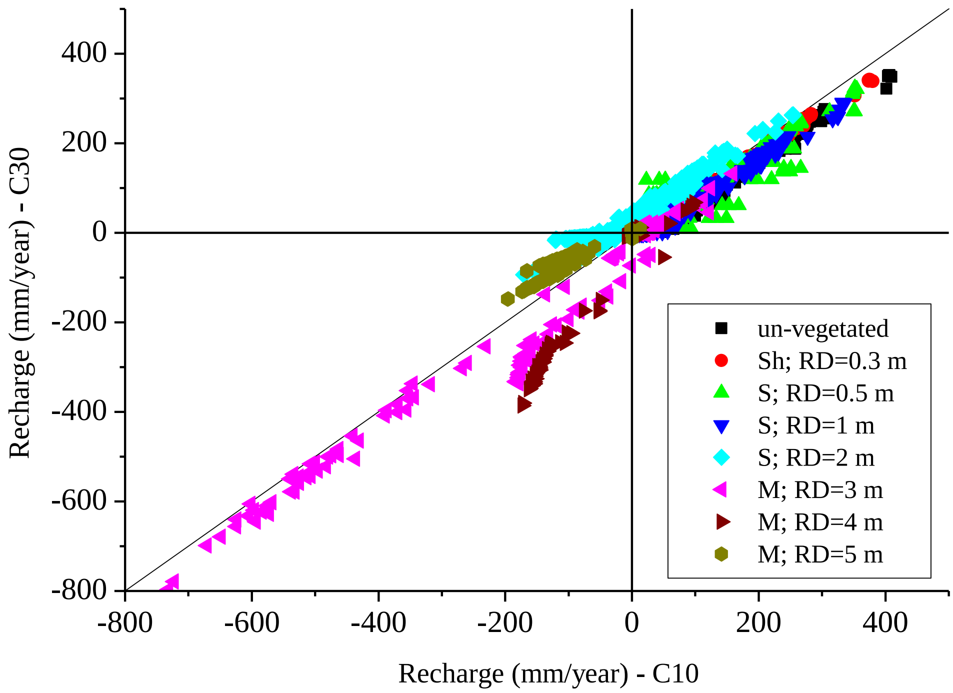

Figure 3.6 – Annual recharge rates for C10 versus C30. Different vegetation types (Sh – shrubs; S – she-oak trees; M – mallee trees) and RD are considered

Figure 3.7 shows a graph of the measured annual rainfall versus calculated annual recharge rates for simple (S10) and complex (C10) profiles. Notably, when rainfall is above 650 mm/year, the annual recharge displays a clear correlation with the annual rainfall (Figure 3.7). This indicates that only under relatively wet conditions, higher rainfall leads to more recharge. When rainfall is less than 650 mm/year, the recharge appears to be independent of the annual rainfall amount, and the scatter of the data points is high. This indicates that when conditions are relatively dry, the episodicity of rainfall events has a major influence on the annual recharge amount, rather than the annual rainfall total, which is in accordance with findings of Barron et al. (2012). Ordens et al. (2012) arrived at the same conclusion based on a comparison of rainfall and measured recharge rates using the WTF method.

The modelling done here provides insight in the importance of the heterogeneity. For example, simulations with mallee trees with RD = 5 m show a stronger correlation between recharge and rainfall for S10 than for C10 for high rainfall rates. This can be explained by the higher permeability of the profile of S10 than C10. That is, the infiltrated rainfall reaches the water table relatively fast during rainfall periods in S10, but in C10, more water is retained and thus transpired by the vegetation after rainfall events, reducing recharge. This control of the vegetation is confirmed by the fact that for un-vegetated conditions, the correlation between rainfall and recharge is virtually the same between C10 and S10.

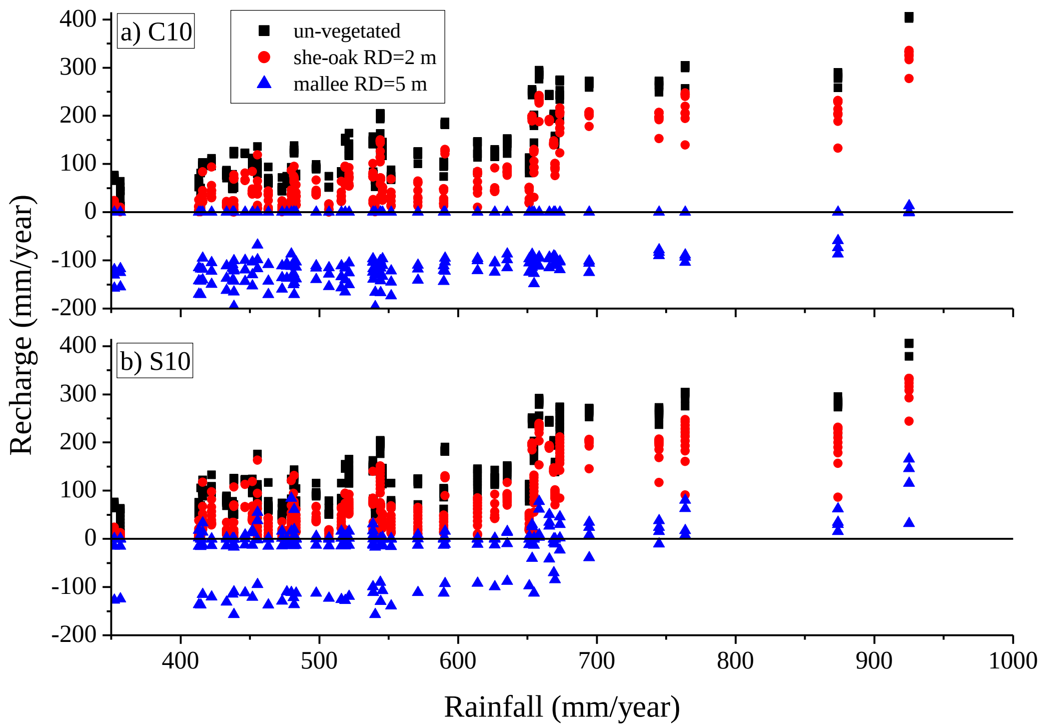

Figure 3.7 – Annual recharge rates versus annual rainfall rates for (a) C10 and (b) S10, and considering un-vegetated conditions, she-oak trees with RD = 2 m or mallee trees with RD = 5 m

Figure 3.8 shows monthly average recharge rates for simple (S10) and complex (C10) lithological profiles, with 5.5 ≤ DWT ≤ 10 m and different vegetation conditions. The differences between simulations C10 and S10 are generally small for un-vegetated conditions, but can become much more pronounced when vegetation is considered. The most remarkable differences are found for DWT values of 6 and 7 m, with mallee trees with RD = 5 m. Figure 3.8 shows that the differences are greatest during the summer months, when the top part of the profile dries out. This is due to (i) more soil water being retained for longer periods of time in C10 than in S10, and (ii) the access of tree roots to groundwater through capillary rise. The combination of these allows transpiration by the vegetation to be sustained for prolonged periods of time. Vegetation is thus the main factor limiting recharge to the system, reinforcing the conclusions drawn from Figure 3.6.

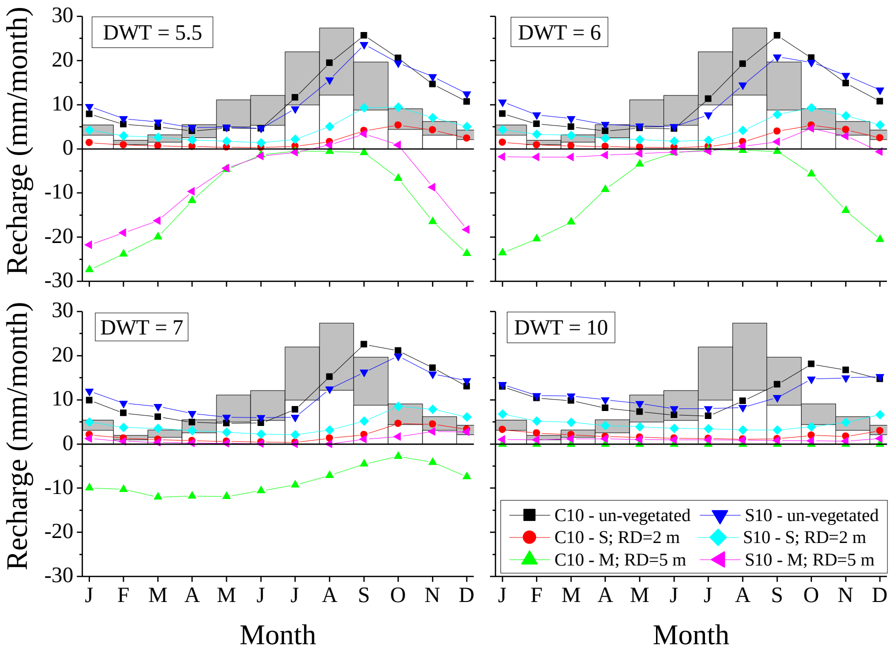

Figure 3.8 – Monthly average recharge rates for C10 and S10. Results are shown for un-vegetated conditions, she-oak trees (S) with RD = 2 m and mallee trees (M) with RD = 5 m; and for 5.5 ≤ DWT ≤ 10 m. The grey columns indicate monthly average recharge based on the WTF method (modified version; Ordens et al., 2012); the top of the grey columns indicate recharge for a Sy of 0.25, while the bottom indicates recharge for a Sy of 0.12

For a DWT of 10 m, hardly any recharge (<1 mm/month) occurs under mallee trees with RD = 5 m. Moreover, the differences between monthly recharge rates are greatly attenuated for large DWT, and the peak recharge occurs significantly later than observed for the field-based monthly recharge rates based on the WTF (Figure 3.5), even for un-vegetated conditions. While the peak recharge occurs earlier with decreasing DWT, and the seasonal trends become more pronounced, the models appear to over-predict the residence time in the unsaturated zone. In what follows, the results of simulations that considered rapid bypass flow will be presented to test the hypothesis that this is a key process in generating recharge in USB.

3.4.2 Bypass flow

The different runoff infiltration scenarios resulted in different annual recharge rates. The differences in annual recharge rates obtained between $C10_0$ and $C10_{1.5}$ are small and mainly noticeable for un-vegetated conditions, where $C10_{1.5}$ produces higher recharges than $C10_0$ (by 11 mm/year on average). In this case, a larger proportion of water escapes evaporation in the top part of the profile by infiltrating through dissolution features, which does not happen in $C10_0$. For vegetated conditions, the water infiltrating through the dissolution features is then available for vegetation use, and therefore the differences in annual recharge rates obtained between $C10_0$ and $C10_{1.5}$ are negligible. $C10_{4.5}$ and $C10_{WT}$ produce higher recharge rates than $C10_0$ for both vegetated and un-vegetated conditions. On average $C10_{4.5}$ produces recharge rates higher then $C10_0$ by: 11 mm/year for un-vegetated conditions; 75 mm/year for she-oak trees with RD = 2 m; and 47 mm/year for mallee trees with RD = 5 m. On average $C10_{WT}$ produces higher recharge rates than $C10_0$ by: 11 mm/year for un-vegetated conditions; 77 mm/year for she-oak trees with RD = 2 m; and 90 mm/year for mallee trees with RD = 5 m. The higher recharge rates occur as a consequence of a higher proportion of water being infiltrated deeper into the profile and therefore escaping evapotranspiration.

Figures 3.9 and 3.10 compare monthly-averaged recharge rates for runoff infiltration scenarios $C10_0$, $C10_{1.5}$, $C10_{4.5}$ and $C10_{WT}$, for $5.5 \leq \text{DWT} \leq 10$ m, and different vegetation conditions (un-vegetated, she-oak trees with RD = 2 m, and mallee trees with RD = 5 m). No significant differences were found between $C10_0$ and $C10_{1.5}$, and this comparison is therefore not shown. The results for scenario $C10_{WT}$ (Figure 3.10) show a recharge peak in July. In these simulations, a larger proportion of infiltrated rainwater reaches the water table without any delay, and therefore the recharge peak is coincident with the rainfall peak. The temporal distribution of monthly recharge based on hydrographs, however, shows that there is a time lag of 1 – 2 months between peak rainfall and peak recharge. Therefore, these modelling outcomes support the conclusion by Ordens et al. (2012) that rapid bypass flow direct to the water table through dissolution features is not a significant contributor to recharge at the basin scale.

With scenario $C10_{4.5}$ (Figure 3.9) there is a clear shift of the recharge peak from mid-winter (July) for shallower water tables (5.5 and 6 m) to late winter to early spring for deeper water tables (7 and 10 m). When the infiltrated runoff bypasses the root zone, or when the DWT is relatively shallow, a clear seasonality in the monthly-recharge rates is observed, but this trend is significantly attenuated by transpiration when the root zone extends below the depth of bypass flow. Great care must be exercised when comparing the WTF-based recharge patterns to the model simulations, as the WTF-based recharge represents the average across USB, and the simulations displayed in Figure 3.9 are representative for the conditions at a particular location. Nevertheless, the results suggest that the observed seasonal recharge pattern may be attributable to a combination of relatively shallow root zones in combination with bypass flow to a depth of a few metres.

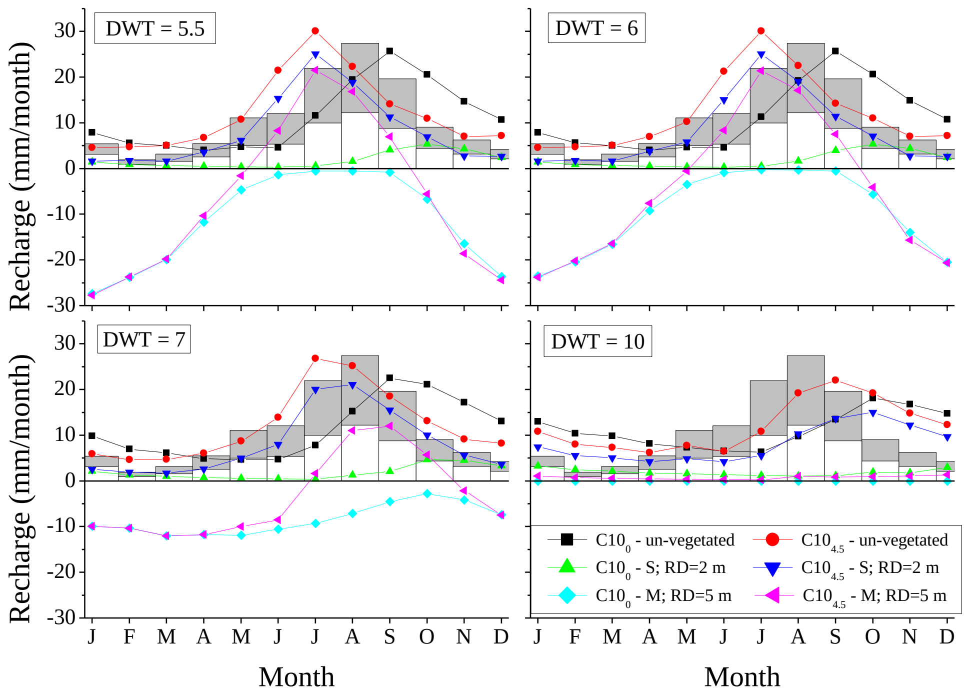

Figure 3.9 – Monthly average recharge rates for C100 and C104.5. The results are shown for un-vegetated conditions, she-oak trees (S) with RD = 2 m or mallee trees (M) with RD = 5 m; and for 5.5 ≤ DWT ≤ 10 m. The grey columns indicate monthly average recharge based on the WTF method (modified version; Ordens et al., 2012); the top of the grey columns indicate recharge for a Sy of 0.25, while the bottom indicates recharge for a Sy of 0.12

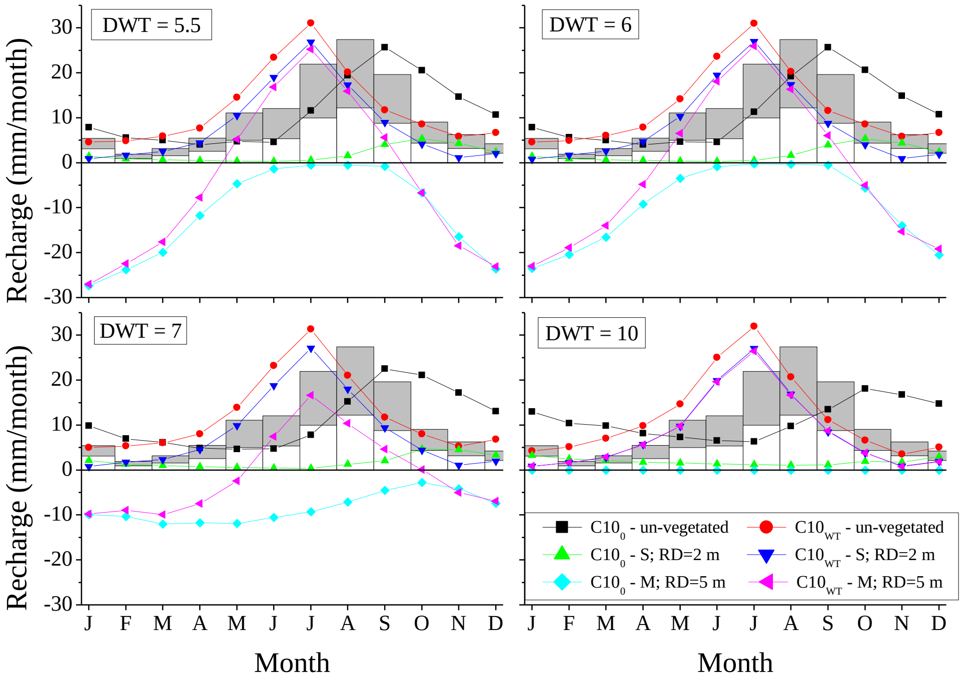

Figure 3.10 – Monthly average recharge rates for C100 and C10WT. The results are shown for un-vegetated conditions, she-oak trees (S) with RD = 2 m or mallee trees (M) with RD = 5 m; and for 5.5 ≤ DWT ≤ 10 m. The grey columns indicate monthly average recharge based on the WTF method (modified version); the top of the grey columns indicate recharge for a Sy of 0.25, while the bottom indicates recharge for a Sy of 0.12

3.5 Conclusions

The one-dimensional simulations of unsaturated zone flow processes to estimate recharge presented here clearly demonstrate the effect of different factors, such as vegetation, lithology, DWT and preferential flow mechanisms, on the magnitude and temporal distribution of recharge. For the hydraulic properties and water retention characteristics adopted, it was found that vegetation type and root zone depth exert the strongest control on the recharge magnitude and timing, which is consistent with other published studies (e.g. Keese et al., 2005; Smerdon et al., 2010; Scanlon et al., 2010). The model simulations showed that the role of lithological complexity is more important when vegetation limits recharge by transpiration than for un-vegetated conditions. This is because the lithological composition determines the amount of water retention and capillary rise, and thus the amount of water available to the vegetation. The depth to water table is important in determining the time-lag and seasonality of the recharge, but annual recharge rates are largely invariant with respect to DWT (Figure 3.5).

This study demonstrates how unsaturated zone modelling is a useful tool to test the effect of different conceptualisations on recharge. It is not possible to distinguish which conceptual models of subsurface lithology are more likely to better represent the system, as they all result in recharge rates that are in the range of long-term, spatially averaged recharge rates by Ordens et al. (2012). Nevertheless, by comparing the timing of the modelled seasonal recharge peak to field data, it appears that partially penetrating bypass flow, in combination with a shallow root zone, is a prerequisite to explain the observed seasonality of the recharge. No single scenario can be identified though, nor is such a scenario likely to exist given the spatial variability within the basin. Therefore, the different conceptual models and the variability in parameters, attributable to both spatial heterogeneity and uncertainties in parameters values, need to be considered when modelling recharge to USB at the basin scale. This is in agreement with what is advocated by Seifert et al. (2008), who affirm that multiple conceptual models need to be considered for assessing the uncertainty of model results.

The spatial variability of some of the factors controlling recharge is known or can be estimated with a reasonable degree of confidence, e.g., the DWT or the vegetation type. For others, such as RD and lithology, the variability is largely unknown. For the purpose of temporal and spatial distributions of recharge at the basin scale, a modelling approach integrating the lessons learnt from this one-dimensional modelling exercise and GIS maps (e.g. Keese et al., 2005) is needed. This will be the focus of future work, because, despite the large uncertainty associated with model-based estimates, numerical simulations of recharge remain the preferred approach for obtaining estimates of the spatial and temporal variability of recharge. The modelling presented in this article can aid in quantifying the uncertainty of model predictions, and can help in guiding data collection efforts. For example, for the USB, a better assessment of the root distribution and vegetation types would increase the reliability of modelled recharge rates.

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4 An application of numerical modelling to extend field-based recharge estimates to the basin scale: Uley South aquifer, South Australia

Abstract

Field-based estimates of recharge are often inadequate for assessing groundwater management options at the basin scale, due to the need to account for spatial and temporal variability in recharge in devising water-use strategies. One-dimensional (1D) unsaturated zone models are commonly advocated to provide temporally and spatially fine resolutions of recharge. Although 1D models are associated with large uncertainties in recharge quantification, they are rarely validated with independent field-based estimates. In this study, field-based methods are combined with numerical modelling to estimate basin-scale recharge to the semi-arid Uley South Basin, South Australia. The 1D unsaturated zone model LEACHM was adopted in an integrated-GIS framework to simulate temporal and spatial variations in rainfall recharge according to depth to water table, topographical slope, substrate characteristics and vegetation type. Variations to the conceptual model that reflect uncertainties associated with complex recharge processes are considered. Results show that the selected combinations of unsaturated zone lithologies and representations of preferential flow produce spatially and temporally averaged recharge rates that fall within the range estimated using the chloride mass balance method. In order to achieve consistency between simulated and field-based recharge timing (inferred from water-table fluctuations), the conceptual model considered the influence of sinkholes by enforcing redistribution of runoff into the unsaturated zone. Because very little unsaturated zone data are available to parameterise and validate the 1D model, the field-based methods proved to be vital to validate the recharge model's predictions. The current study successfully used field-based estimates to constrain the model results and as such, imparts improved confidence in both methods for basin-scale recharge estimation.

4.1 Introduction

An assessment of the spatial and temporal distributions of groundwater recharge at the basin scale is often essential for effective groundwater management purposes (e.g. Healy, 2010; Werner et al., 2011). Field-based methods are generally inadequate for such purposes because they do not provide insight into recharge variability in the time and spatial domains. Hence, numerical models are often employed as a means of meeting the information needs of management decision-making, and for groundwater modelling input. Keese et al. (2005) advocate the use of one-dimensional (1D) unsaturated zone modelling to calculate recharge, because it is the only approach capable of predicting future recharge patterns, while enabling examination of the relative importance of various controlling factors on recharge (e.g. climate, vegetation, unsaturated zone lithology, topography). Nevertheless, recharge cannot be properly estimated by a single method (e.g. Flint et al., 2002; Scanlon et al., 2002a; Healy, 2010). The application of multiple techniques can increase the reliability of recharge estimates (Scanlon et al., 2002a), and assist in identifying possible errors when inconsistent results are obtained (Healy and Cook, 2002). Modelling and field-based approaches represent two different philosophies towards estimating groundwater recharge. Unsaturated zone modelling simulates the key atmosphere-vegetation-unsaturated zone processes to calculate recharge. Field-based approaches are usually indirect and represent the integration of recharge pathways, and often use groundwater observations to imply the response of the subsurface to influxes.

Keese et al. (2005) report that the primary difficulties associated with 1D recharge modelling are data availability and model parameterisation. Recharge rates obtained using 1D models can be highly uncertain due to difficulties in characterising hydraulic conductivity across extensive areas, and given nonlinearities between hydraulic conductivity, water content and matric potential (Scanlon et al., 2002a). Other primary sources of uncertainty associated with 1D recharge modelling include boundary condition specification (e.g. Carrera-Hernández et al., 2012), profile discretisation (e.g. Carrera-Hernández et al., 2012), model conceptualisation (e.g. Ordens et al., 2014) and preferential flow representation (e.g. Köhne et al., 2009). The accuracy of field-based approaches depends predominantly on the spatial and temporal resolution of sampling, measurement error, and uncertainties in up-scaling (Healy, 2010). However, validation of modelled recharge rates by critically combining field-based methods and 1D modelling, and thereby obtaining independently verified recharge estimates, is not common practice (e.g. Jyrkama et al., 2002).

Previous studies that compare 1D modelling and field-based recharge estimates include those of Keese et al. (2005), Anuraga et al. (2006), Scanlon et al. (2010) and Ireson and Butler (2011). Keese et al. (2005) modelled recharge across the State of Texas, USA (some 700,000 km^2) using a 1D–GIS framework. The match between modelling results and field-based estimates, based on the groundwater and/or unsaturated chloride mass balance (CMB) approaches applied to seven of the thirteen modelled regions, was mixed. The modelling results compared well with CMB estimates in three regions, whereas in four other regions, the modelling predictions were significantly lower than field-based values. They explained the discrepancies between the two approaches as a result of (i) focused recharge (originated from playas) occurring in two of the regions, which could not be captured by the 1D model (i.e. only matrix flow was considered); and (ii) field measurement bias towards units of higher permeability. Scanlon et al. (2010) used 1D modelling to test the plausibility of recharge estimates obtained using the unsaturated zone CMB approach in an arid sedimentary aquifer in the Thar Desert, India. Different scenarios were used to analyse the sensitivity of modelled recharge to different vegetation parameters. Results were only consistent with the CMB recharge estimates for one of the rooting depths (RD) (i.e. 1 m) considered in the model. They stressed that the model had many limitations, namely the lack of site-specific data (e.g. hydraulic parameters, vegetation characteristics) and the short simulation period (i.e. 8 years). Anuraga et al. (2006) used 1D modelling linked to a GIS to calculate recharge to the 177 km^2 semi-arid Bethamangala sub-watershed, India. They compared model predictions with recharge estimates obtained using the water-table fluctuation (WTF) method. The two methods produced average recharge rates with a discrepancy of up to 25%. Anuraga et al. (2006) partly attributed the mismatch to uncertainty in the specific yield (S_y) value that is required by the WTF method. These comparisons between field-based and modelling approaches only considered temporally averaged recharge rates, and did not take into account the temporal variability of recharge within the modelled regions. Additionally, Ireson and Butler (2011) compared 1D model-based timing of recharge (considering preferential flow through fractures) with water-table fluctuation data in chalk aquifers in the Pang and Lambourn catchments, UK. They found that the modelled timing of recharge was reasonably consistent with water-table fluctuations, although lags between recharge pulses and water-table rises were observed. They speculated that differences were caused by lateral flow processes within the saturated zone. These studies highlight an overall lack of agreement between model- and field-based recharge estimates.

There are few critical evaluations of discrepancies between field-based and modelling methods. Even when recharge time-averages are in agreement between approaches, often no consideration is given to the consistency between modelled and observed timings of recharge. There is a need for further research into the causes of discrepancies between different recharge estimation approaches applied to basin-scale investigations. Also, improved modelling tools and well-documented case studies to better combine field-based and modelling approaches are required.

This study extends upon the previous work of Ordens et al. (2012) and Ordens et al. (2014) by combining 1D unsaturated flow modelling and field-based methods to provide predictions of basin-scale, spatial and temporal recharge distributions for the Uley South Basin (USB), South Australia. As with many arid and semi-arid regions, there are limited alternative water supply options in the Eyre Peninsula, and therefore recharge estimations are vital. Presently, USB is managed entirely on recharge estimates (Werner at al., 2011). Consequently, there is a need for a predictive tool capable of simulating temporally and spatially variable recharge to USB.

Regional-scale recharge estimates are obtained by combining previous outcomes from a (i) field-based recharge study (Ordens et al., 2012), (ii) a previous analysis of the influence of USB conceptualisation on 1D recharge modelling (Ordens et al., 2014), and (iii) spatial distributions of depth to water table (DWT), vegetation type, substrate type and topographical slope. Due to the hard nature of the terrain in USB (limited soils over extensive calcrete areas), there is a scarcity of data (e.g. unsaturated zone monitoring) to parameterise and calibrate the 1D model. Therefore, the model's outputs are compared to the integrated signals of groundwater-based field-estimates of recharge in USB.

4.2 Study Area

The physical setting of USB has been described by Harrington et al. (2006), Werner et al. (2011), Ordens et al. (2012) and Ordens et al. (2014), and as such, we review only the aspects that are essential to understanding USB's recharge process. USB is a 129 km^2 topographically closed sedimentary basin, which is bounded to the southwest by coastal cliffs that overlook the Southern Ocean, and by topographic rises to the northwest, northeast and southeast (Figure 4.1). Recharge to USB occurs via rainfall infiltration into the unconfined Quaternary Limestone aquifer (QL), which overlays Tertiary sequences of clay and sand. Inter-aquifer recharge into USB from surrounding aquifer systems and seawater intrusion may also be occurring, although this is not well constrained in previous studies (Harrington et al., 2006; Werner et al., 2011). Discharge from the basin is due to evapotranspiration, groundwater abstraction and groundwater flow to the sea (Werner et al., 2011; Ordens et al., 2012). The QL consists of a heterogeneous combination of unconsolidated to poorly consolidated aeolian sediments and calcrete of varying hardness and porosity, with karstic features in places. A thin layer of calcareous or loamy soil overlays the QL (Harrington et al., 2006) that is often thicker under vegetated sites, and is absent in some areas (where calcrete is exposed). Thousands of dissolution features, including sinkholes, exist across the basin, and their diameter varies from a few millimetres to several decimetres (Twidale and Bourne, 2000). The QL in the north-western part of USB is overlain by calcareous sand dunes, in which no soils are present. USB presents a heterogeneous distribution of vegetation, and the major vegetation types include (i) mallee trees (*Eucalyptus diversifolia* and *Eucalyptus gracilis*), (ii) she-oak trees (*Allocasuarina verticillata*), (iii) shrub species (e.g. *Leucopogon parviflorus* and *Melaleuca lanceolata*), and (iv) sparse grass (during the wetter months, otherwise absent).

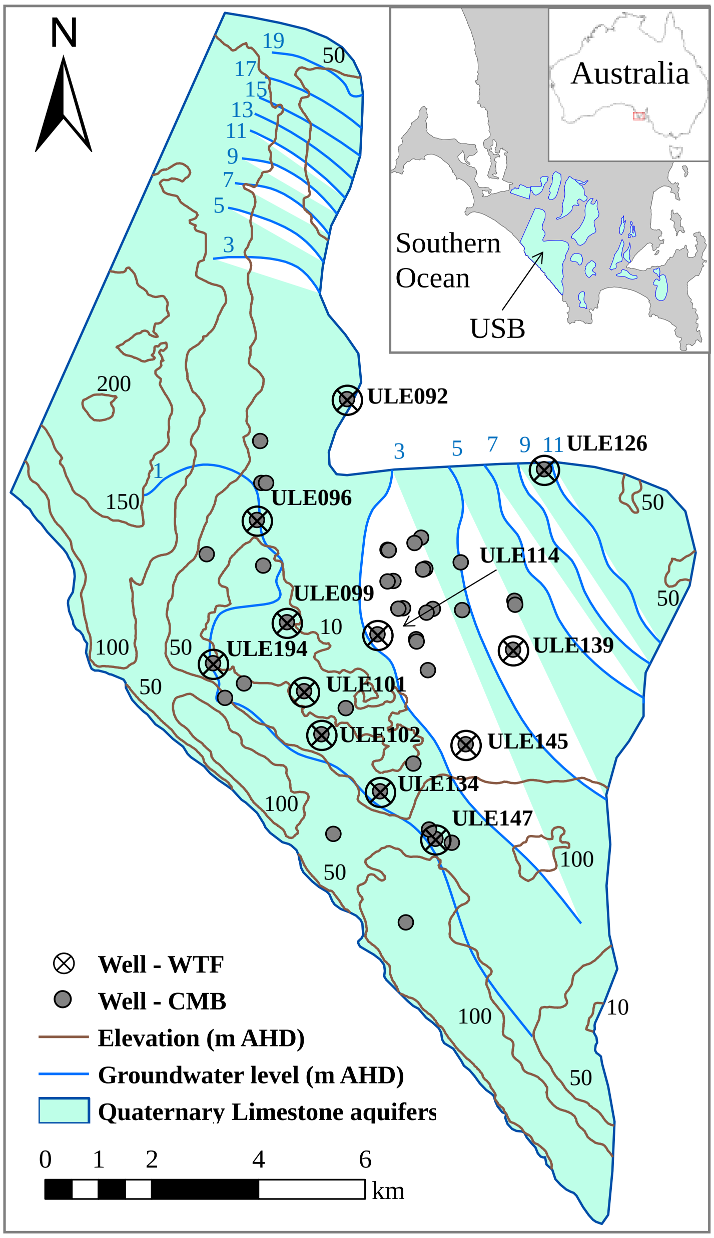

Figure 4.1 – Map showing the extent of USB (as defined by Harrington et al. (2006) as the limit of saturated QL), the location of wells used for WTF (labelled) and CMB estimates of recharge, groundwater levels (from 2007 measurements) and the topographic elevation. AHD is Australian Height Datum, roughly equal to mean sea level

The semi-arid climate of USB is characterized by cool, wet winters and hot, dry summers (Zulfic et al., 2007). The average annual rainfall is 560 mm/year (measured or interpolated), the average FAO56 reference evapotranspiration (Allen et al., 1998) is 1084 mm/year (interpolated), and the average pan evaporation is 1547 mm/year (interpolated). Climate data for the study area were obtained from DSITIA's (2013) Patched Point Dataset (PPD) for the Big Swamp monitoring station (18017), located 18 km northeast of the basin, where daily data (either measured or interpolated) are available dating back to 1889. The patched point dataset data set was chosen in preference to alternative Data Drill (DSITIA, 2013) because the latter showed severe inconsistencies between measured and interpolated rainfall data. The Data Drill dataset includes many more days of rain than were observed.

The recharge processes in USB were investigated by Ordens et al. (2012) using various field observations (hydrochemical, isotopic and water level data) and by Ordens et al. (2014) based on 1D unsaturated zone modelling. Ordens et al. (2012) suggest that recharge in USB can be conceptualised as follows: (i) rainfall is partitioned into evapotranspiration and groundwater recharge at the basin scale, as there is no runoff to the sea due to the closed nature of the basin; (ii) surface runoff is highly ephemeral in USB, occurring only after prolonged rainfall events and persisting for tens to hundreds of metres before infiltrating through preferential flow features (e.g. sinkholes, fractures) (Harrington et al., 2006), and hence the redistribution of rainfall across the land surface is likely to be minimal over kilometre scales; (iii) preferential flow features mainly distribute water deeper in the unsaturated zone (rather than directly to the water table), allowing access by vegetation, although some direct recharge could not be excluded; (iv) a substantial proportion of rainfall occurring in dryer months may be subjected to complete evaporation at the surface; and (v) unsaturated zone water and groundwater are subject to transpiration more so than evaporation.

Ordens et al. (2014) concluded that (i) preferential flow processes need to be accounted for, because the simulation of matrix flow only (i.e. neglecting preferential pathways) produced erroneous temporal recharge trends based on the WTF-based timing of recharge; (ii) direct recharge from the surface to the water table via sinkholes is unlikely to provide the dominant source of recharge to the basin; (iii) different complexities of the unsaturated zone lithology can result in significantly different recharge rates, and because lithological complexity and composition are likely to be highly variable across the basin, both simple and complex representations of subsurface lithology need to be considered in basin-scale recharge simulations; and (iv) vegetation distribution and parameterisation are important drivers of modelled recharge rates (because evapotranspiration is the only sink of surface water from the system).

Ordens et al. (2012) applied the CMB method to estimate temporally and spatially averaged recharge to USB. A range of 52 to 68 mm/year was obtained depending on assumptions made about atmospheric chloride deposition. The upper bound CMB estimates of Ordens et al. (2012) correspond to atmospheric chloride (rainfall and dry) deposition taken at the well locations, while the lower bound correspond to atmospheric deposition taken at the up-gradient boundary of the aquifer (inferred as the uppermost possible recharge location for the wells from CFC age dating and aquifer hydraulic properties). They found that CMB recharge rates vary between well locations, from 37 to 115 mm/year, and that higher CMB recharge rates were related to areas of limited vegetation. The authors highlight several sources of uncertainty in the CMB estimates and some potential errors were approximated, including: (i) the timing of sampling (8-15% error), (ii) groundwater sampling depth (<3% error); and (iii) the effect of vegetation on the atmospheric chloride deposition. The influence of vegetation on chloride deposition may result in a substantial increase in estimated recharge rates based on some preliminary field measurements. Further work is required to quantify this uncertainty, which is the subject of a concurrent investigation (Bresciani et al., 2013).

Ordens et al. (2012) obtained annual recharge rates of 47 to 129 mm/year using the WTF approach, which was modified to account for seasonal pumping. This range reflects uncertainty in specific yield (S_y) values (0.12 to 0.25). The WTF results indicated that recharge to USB occurs predominantly during July to September (Ordens et al., 2014). Recent re-analysis of USB's lithostratigraphy showed that three wells used by Ordens et al. (2012) are located in the Tertiary Sand aquifer (TS), which underlies the QL. Revised CMB and WTF estimates excluding those three wells result in a modified range of 53 to 70 mm/year and 47 to 128 mm/year, respectively, and these will be adopted herein.

4.3 Methodology

4.3.1 Model description

Recharge modelling adopted the modified version of LEACHM (Hutson, 2005) proposed by Ordens et al. (2014) that allows for re-distribution of surface runoff into the unsaturated zone, as discussed below. LEACHM was linked to a GIS framework, referred to as LEACHMG herein, thereby incorporating spatial distributions of input data (through raster information) into the recharge modelling approach. LEACHMG uses a finite-difference approach to solve a 1D form of Richards' equation:

$$\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z}\left(K(\theta)\frac{\partial H}{\partial z} \right) - S \tag{4.1}$$

where θ is soil moisture content [L^3L^{-3}], K is hydraulic conductivity [LT^{-1}], H is hydraulic head [L], t is time [T], z is depth [L] (positive downwards), and S is a source/sink term [T^{-1}]. The two-part modification of Campbell's (1974) K-θ-h functions (where h [L] is pressure potential) was used to describe hydraulic relationships of the unsaturated zone (Hutson and Cass, 1987).

The boundary condition at the land surface accounted for rainfall and reference evapotranspiration. In LEACHMG, reference evapotranspiration is divided into potential transpiration and potential evaporation depending on the foliage projection cover (FPC), which ranges from 0 (no transpiration) to 1 (no evaporation). LEACHMG defines vegetation in terms of FPC, rooting depth (RD) and duration of growth (i.e. periods of active transpiration). Surface runoff is generated in LEACHMG by partitioning rainfall into runoff and potential infiltration according to the slope-based curve number approach of Williams (1991). This approach is based on empirical relationships that consider both the hydrologic characteristics of the surface and the antecedent soil water content at the near-surface (USDA, 1986).

Additional runoff can be generated in LEACHMG under conditions of (i) saturation-excess runoff (i.e. storage availability in the soil is exceeded), and (ii) infiltration-excess runoff (i.e. the infiltration capacity of the soil exceeds rainfall). As per Ordens et al. (2014), preferential infiltration is considered, whereby surface runoff can be transmitted directly into the unsaturated zone at any user-specified depth through the S term in Equation (4.1), thereby accounting for the rapid bypass flow via sinkholes or other preferential flow features. Runoff water is assigned to the segment immediately above if saturation is reached at a given node. Further details of the model are described by Ordens et al. (2014).

4.3.2 Model application

The current application of LEACHMG extends the methodology of Ordens et al. (2014) by developing a basin-scale assessment of recharge using raster information on distributions of substrate type, vegetation type, DWT and slope across the basin (Figure 4.2). The simulation period was from January 1957 to December 2007. The model cycled through this period twice; the first cycle was used as an equilibration period to generate initial conditions; the second cycle was used to generate recharge predictions.

Rainfall and reference evapotranspiration (modelled pan evaporation data) used for the upper boundary condition were derived from historic daily weather data from Big Swamp weather station (DSITIA, 2013). A constant positive pressure potential was imposed as a lower boundary condition to represent the (fixed) water table. Ordens et al. (2014) showed that implementing a fixed-depth water table, similar to the approach of Smerdon et al. (2008), was more appropriate than the unit-gradient boundary condition that was adopted by Keese et al. (2005) because the DWT spatial variability influences recharge modelling across the basin. Various DWT values, which were based on average monthly piezometer measurements and digital elevation data (DED; Geoscience Australia, 2013), were considered in specifying the pressure potential applied to the lower boundary (Figure 4.2c). Lower boundary conditions used in the LEACHMG simulations correspond to DWTs of 5, 7 and 10 m. Land surface slopes in USB, shown in Figure 4.2d, were obtained from

Geoscience Australia's (2013) DED. These were used in curve-number (Williams, 1991) calculations of runoff, and correspond to slope percentages of 1, 3, 5, 10 and 15. Figure 4.2b is adapted from a map of vegetation types by Ordens et al. (2012) and was used as the vegetation distribution for LEACHMG in the current study. Similarly, Figure 4.2a is adapted from a map of substrate types from the same study, and was used as the substrate distribution in the modelling.

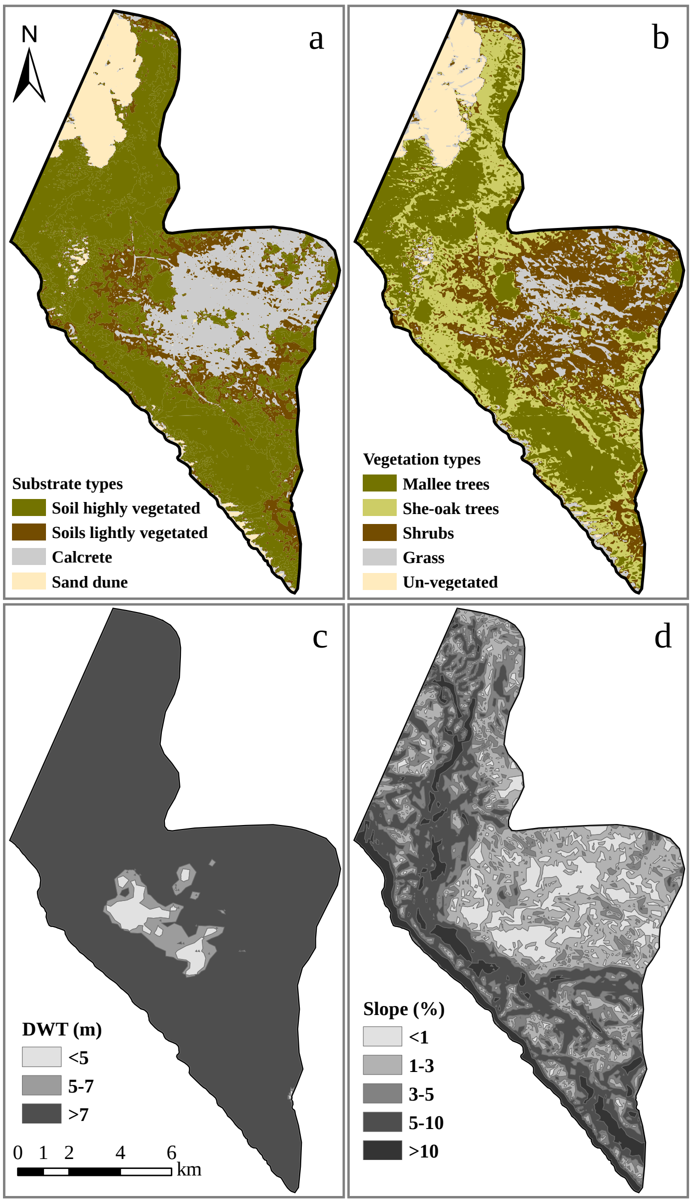

Figure 4.2 –Spatial distribution of (a) substrate type, (b) vegetation type, (c) DWT and (d) topographical slope, used as LEACHMG inputs. Maps (a) and (b) are adapted from Ordens et al. (2012). All raster files have a pixel size of 70 m

The parameters used to represent vegetation in this study are similar to those adopted by Ordens et al. (2014). FPC values were obtained using the point quadrat technique (e.g. Specht and Morgan, 1981), and are as follows: 0.5 for mallee trees, 0.3 for she-oak trees and 0.2 for shrubs. A low FPC value of 0.1 was adopted for sparse grass in the absence of field data (e.g. Tunstall, 2008). Mallee trees, she-oak trees and shrubs were considered to be active throughout the year, while grass was present from May to October. Because no specific information exists on RD in USB, Ordens et al. (2014) considered a range of RD values based on literature guidance (Canadell et al., 1996; Wright and Ladiges, 1997; Sun and Dickinson, 1995). RD values of 5 m for mallee trees, 2 m for she-oak trees, and 0.3 m for shrubs were taken as upper values from (Ordens et al., 2014). A RD of 0.1 m was used for grass.

The unsaturated zone profiles were defined to a depth of 10 m, using a constant node spacing of 0.1 m. Various plausible lithological profiles, which control the vertical distribution in unsaturated zone hydraulic properties, were tested in the modelling analysis. The hydraulic properties of different lithological layers were characterised by Hutson and Cass (1987) *a* and *b* retention function parameters, bulk density, and *K* at a defined matric potential, as used by Ordens et al. (2014). Here, we used the map of substrate types from Ordens et al. (2012) (Figure 4.2a), and attributed the profiles of 10 cm top soil to the lightly vegetated areas, and the profiles of 30 cm top soil to the highly vegetated areas. Both simple and complex conceptualisations of subsurface lithology were adopted in our basin-scale recharge simulations. The lithological structures adopted to represent the unsaturated zone profiles are as follows: (i) complex profiles (from top to bottom): 10 or 30 cm of top soil, 1 m of calcrete, 1 m of sand, and 1 m of clay (the sequence calcrete-sand-clay is repeated to the bottom of the profile); (ii) simple profiles (from top to bottom): 10 or 30 cm of top soil, underlain by 1m calcrete layer and then sand to the bottom of the profile. An area where calcrete is exposed at the surface exists in USB (Figure 4.2a), and therefore we also considered profiles where top soil is absent (i.e. the upper profile is solely calcrete). An area of the basin composed of sand dunes (Figure 4.2a) was represented by a homogeneous sand profile. The curve numbers corresponded to values considered appropriate for semi-arid rangelands and were chosen to reflect differences in the upper subsurface lithology (USDA, 1986): a curve number of 80 was assigned to 30-cm top soils, a curve number of 85 was assigned to 10-cm top soils, and a curve number of 90 was assigned when calcrete was at the surface; this excludes the sand dune area, where only matrix flow is considered given the high infiltration rates of homogeneous sand.

Only scenarios where preferential flow is considered are modelled, following the results of Ordens et al. (2014), who eliminated matrix-flow only as likely recharge mechanism at the regional scale. Although they concluded that direct recharge to the water table is not likely to be significant at basin scale, we include that scenario for completeness. Scenarios include runoff redistribution to 1.5 m, 4.5 m or 7.5 m (this scenario applies only to situations where the water table exceeds 7.5 m depth, otherwise the runoff redistribution depth is 4.5 m), and directly to the water table. Eight combinations of lithology and runoff redistribution were considered: $S_{1.5}$, $C_{1.5}$, $S_{4.5}$, $C_{4.5}$, $S_{7.5}$, $C_{7.5}$, S_{WT} and C_{WT} where S and C refer to simple and complex profiles, respectively, and subscripts identify the runoff redistribution depth.

4.3.3 Comparison of modelled and field-based recharge rates

The modelled recharge rates were compared to field-based estimates (CMB and WTF results) presented by Ordens et al. (2012). CMB recharge estimates were compared to time-averaged estimates from the model thus providing insight into spatial recharge distributions. Conversely, results from WTF analyses were used to assess the timing rather than the magnitude of recharge, due to the large uncertainty in S_y (that effect WTF-based magnitudes). The catchment contributing area of each groundwater sampling site needed to be approximated to allow comparison between LEACHMG and field-based methods. A 1-km radius circular area, centred at each well, was adopted to extract area-averaged recharge from the model, which is similar to the approach of Anuraga et al. (2006). This is a somewhat crude method of recharge averaging in the absence of more complicated groundwater flow calculations (e.g. from a distributed flow model) to predict areas of recharge capture (for CMB) or recharge response (for WTF) for each well with greater accuracy. Because no field-based recharge estimates exist for the sand dune area, the modelled recharge rates obtained for the whole basin cannot be readily compared to field-based methods. Furthermore, the sand dune area may be located (at least partially) outside of the hydrogeological boundaries of USB, based on a concurrent study (Knowling et al., in prep.). Therefore, basin-averages that both include and exclude the sand dune area are reported.

Although 1D modelling and WTF represent recharge as the vertical movement of water through the unsaturated zone, there are important differences between what is considered recharge between the model- and WTF-based approaches. The former represents recharge as the arrival of unsaturated zone flow to the water table. In contrast, the latter uses water-table response (i.e. water-table rise) to infer recharge. Aside from time-lag issues, the WTF approach also masks recharge occurring during water-table decline periods, if recharge is less than discharge (Healy and Cook, 2002). This issue is particularly relevant for high-*K*, coastal aquifers, such as USB, where coastal discharge is an important component of the water balance (Werner et al., 2011). We presume that the high transmissivity of USB results in relatively short water-table response times. Given that water-level measurements are recorded on a monthly basis, time-lag inconsistencies between modelled and WTF recharge estimations are likely to be masked in any case.

The Nash-Sutcliffe coefficient (NSC) (Nash and Sutcliffe, 1970) was used to evaluate the modelled timing of recharge, both for monthly and annual recharge rates. WTF and modelled recharge were normalised by dividing by their respective averages to focus the analysis on timing rather than magnitude discrepancies. The NSC coefficient is given as:

$$NSC = 1 - \frac{\sum_{i=1}^{n}(o_i - p_i)^2}{\sum_{i=1}^{n}(o_i - \sigma)^2} \tag{4.2}$$

where o_i and p_i are normalised WTF and modelled recharge at time step i, respectively, and σ is the average normalised WTF recharge.

4.4 Results and Discussion

4.4.1 Temporally averaged recharge

Table 4.1 lists simulated spatial and temporal average recharge rates from the eight LEACHMG scenarios, which range from 77 to 89 mm/year (14–16% of rainfall) if the sand dune area is considered, and 61 to 74 mm/year (11–13% of rainfall) if the sand dune area is excluded.

Table 4.1 – Spatially and temporally averaged recharge rates from six modelled scenarios

	Recharge (mm/year)	
	Total basin	Excluding sand dune area
$\mathbf{S_{1.5}}$	82	67
$\mathbf{C_{1.5}}$	77	61
$\mathbf{S_{4.5}}$	84	69
$\mathbf{C_{4.5}}$	79	64
$\mathbf{S_{7.5}}$	89	74
$\mathbf{C_{7.5}}$	84	69
$\mathbf{S_{WT}}$	89	74
$\mathbf{C_{WT}}$	87	72

Table 4.1 shows that, at the basin scale and excluding the sand dune area, simulations considering runoff infiltration to a depth of 1.5 and 4.5 m yield average recharge rates that differ by only 2 mm/year for simple lithological profiles, and 3 mm/year for complex lithological profiles. The redistribution of runoff into the unsaturated zone to a depth of 7.5 m yield average recharge rates 5 to 7 mm/year higher. Simulations assuming direct recharge to the water table result in similar averaged recharge rates for simple profiles, and in averaged recharge rates 3 mm/year higher

for complex profiles. Simulations of simple lithological profiles result in average recharge of 5 mm/year higher than simulations of complex lithological profiles for the simulations redistributing surface runoff into the water table; for the scenarios of direct recharge to the water table the difference was 2 mm/year. These trends are consistent with the findings of Ordens et al. (2014), in that complex lithological profiles and shallow runoff infiltration depths are more conducive to higher evapotranspiration rates due to the enhanced availability of unsaturated zone water (and therefore less recharge). As such, greatest basin-scale average recharge estimates are achieved using simple lithological profiles and deep redistribution of runoff.

Furthermore, model-based estimates are mostly in the range of the CMB-based results of 53 to 70 mm/year from Ordens et al. (2012). This outcome was obtained without undertaking model calibration, albeit parameter value selection in the current study was informed by conceptual model testing and sensitivity analyses undertaken by Ordens et al. (2014). Nonetheless, the consistency between model- and field-based estimates promotes confidence in the model's representativeness of the unsaturated zone water balance and basin-scale recharge. Given that ranges in average recharge are produced from both modelling and field-based methods, it is not possible to establish which of the simulations are more likely to better represent the recharge processes within the basin.

The comparison of the simulations and CMB results for individual wells (i.e. using a circular area with a 1 km radius centred at each well for extracting modelled recharge) is presented in Figure 4.3. The relationship between modelled and CMB recharge estimates for individual wells is clearly weak. A well (located in a highly vegetated, shallow water table area) for which negative recharge (-111 to -76 mm/year) was predicted by the model was omitted from Figure 4.3 to avoid graph scale distortion. Upper bound CMB recharge for this well is 95 mm/year (CMB recharge estimates cannot be negative). LEACHMG simulations produced, on average, higher recharge rates by 5 to 22 mm/year (depending on the simulation) than the CMB estimates. Figure 4.3 shows that LEACHMG produced higher recharge rates than the CMB method in about 57% of the wells, except in simulation $S_{7.5}$ where LEACHMG was higher in 68% of the wells.

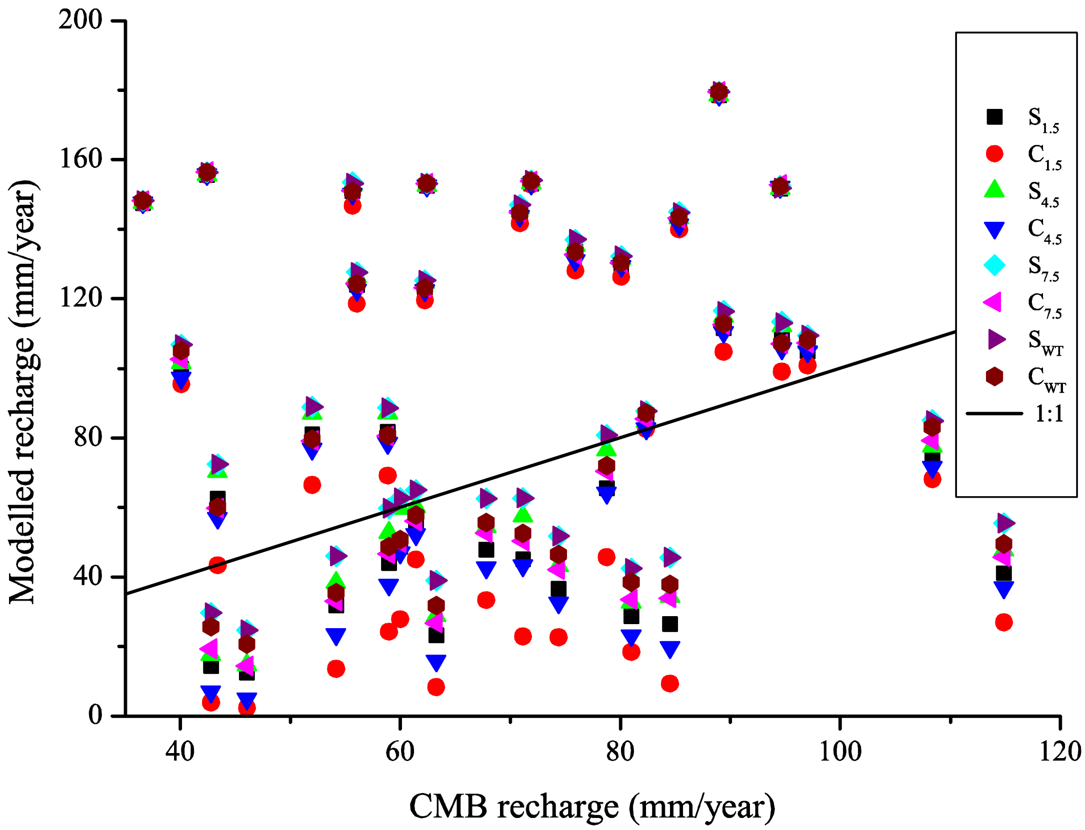

Figure 4.3 –Recharge rates obtained with CMB (upper bound) and LEACHMG

Figure 4.4 shows a spatial distribution map of modelled recharge for $S_{7.5}$ and $C_{7.5}$ simulations (considered the best simulations of recharge as discussed below), compared to CMB recharge. High modelled recharge rates, of approximately 250 mm/year, are obtained in the sand dune area, as expected, due to (i) the absence of vegetation and (ii) the lithologic composition. The area where calcrete is exposed and limited vegetation exists also produces high recharge rates, in the range 100 to 200 mm/year. Figure 4.4 shows areas of negative modelled recharge, which occur in regions where mallee vegetation and shallow water tables coincide, producing high transpiration rates because the trees have access to the water table via the capillary fringe. These findings are consistent with those of Swaffer et al. (2013), who found evidence of negative recharge in USB under areas of mallee and she-oak trees and shallow water tables, albeit their study was limited to only two sites and a period of 24 months.

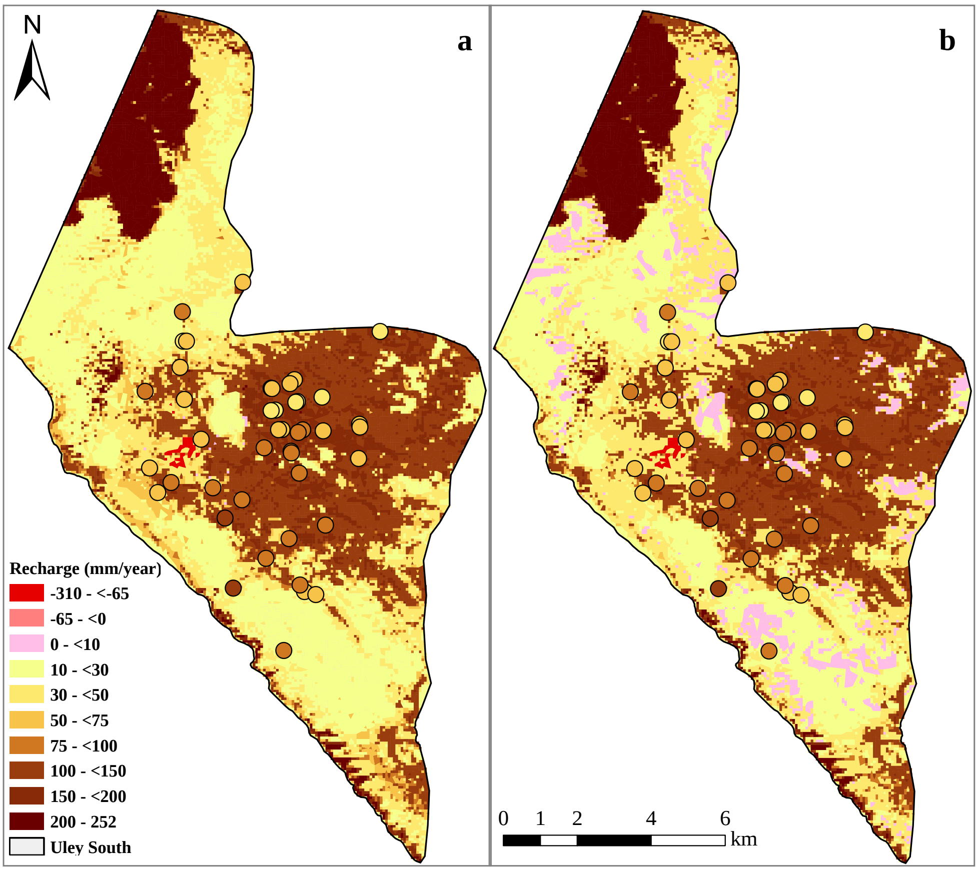

Figure 4.4 – Spatial distribution of modelled temporally averaged (a) $S_{7.5}$ and (b) $C_{7.5}$ recharge rates (mm/year) and upper bound CMB recharge rates (mm/year). Simulated and CMB recharge rates use the same colour scale

Similarly to Figure 4.3, Figure 4.4 does not reveal a strong match between CMB and simulated recharge rates for individual wells. Figures 4.3 and 4.4 highlight that the comparison of LEACHMG and CMB results, taking into account the spatial distribution of wells and recharge, shows complicated relationships. A key challenge is the definition of recharge zone for individual wells, which requires consideration of (i) the aquifer hydraulic properties, which are highly heterogeneous; (ii) the depths and lengths of well screens, and the associated depths and methods of sampling; and

(iii) mixing within the aquifer. These issues, in addition to short-comings in well construction details, make it difficult to attribute specific chloride measurements to well-defined recharge areas.

Figure 4.4 shows a general pattern of wells with higher CMB recharge rates down-gradient of areas of higher simulated recharge. This is most likely a consequence of the recharge area of the wells being located up-gradient of the wells, at least based on time-averaged groundwater head gradients (Figure 4.1). An attempt to account for this was made by shifting the 1-km radius circles (used for assigning LEACHMG recharge to individual wells) in the up-gradient direction. Distances at which recharge zones were located up-gradient were approximated from average groundwater flow velocities and time-scales based on groundwater age (averaging 11 to 14 years as per Ordens et al., 2012). Considering plausible parameter ranges, up-gradient distance to recharge zones varied from 900 m to the boundary of the aquifer. The results were mixed, while in some cases the match between modelled and CMB estimates improved, in some others it did not. There was no evident overall improvement in the results. This requires further investigation. A groundwater flow model of the basin is currently being developed (Knowling et al., in prep.), and as such, is expected to assist in identifying recharge zones for wells, allowing for an improved comparison between modelled and CMB recharge rates on a well-to-well basis.

4.4.2 Monthly and annual recharge rates

The timing of recharge is an important aspect of the model's ability to reproduce recharge in the basin. Figure 4.5 shows the monthly average recharge rates (normalised) from LEACHMG simulations (considering the 1-km circle centred in each well) and WTF. Recharge according to the WTF method occurs mainly in July (16.8%), August (20.6%) and September (14.9%), with the lowest recharge occurring in February (1.6%). Months of higher recharge immediately follow the high rainfall (peaks in July) and low evapotranspiration (minimum in June) season, indicating a delay in peak recharge compared to peak rainfall. Simulations that better resemble the timing and relative magnitude of WTF estimates are the ones considering deeper runoff infiltration into the unsaturated zone ($S_{7.5}$ and $C_{7.5}$). Simulations that adopt shallower runoff infiltration depths ($S_{1.5}$, $C_{1.5}$, $S_{4.5}$ and $C_{4.5}$) produce significant differences in recharge timing when compared to the WTF results. For example, the highest recharge rates occur during September–December, which is considerably later than the WTF trends. Also, the lowest recharge occurs during the period April–July in simulations $S_{1.5}$ to $C_{4.5}$, at a time when WTF-based recharge estimates are relatively high. Clearly, these four simulations predict a slower recharge process than what is indicated by the WTF and thus supports the presence of deeper runoff infiltration depths. On the contrary, the scenarios were direct recharge to the water table is considered (S_{WT} and C_{WT}) predict a faster recharge process then the WTF estimates. A peak of recharge is observed in June (20%) followed by a rapid recharge decrease, coincident with the rainfall peak. This is consistent with the findings of Ordens et al. (2012) (based on hydrochemistry data) and Ordens et al. (2014) (based on modelling) that preferential flow is not likely to conduct recharge directly to the water table, but to redistribute surface runoff into the unsaturated zone. Despite of a good match between $S_{7.5}$ and $C_{7.5}$ simulated and WTF timing of recharge, some differences can be identified. For example, whilst a rapid decline in WTF-based recharge after July–September is observed, both simulations predict significant recharge after this winter period. For example, the WTF recharge rates show a sharp drop from September to October (7.7%), while the $S_{7.5}$ and $C_{7.5}$ recharge rates show a much less pronounced decreased (2.3 and 2.7%). This is likely caused by WTF not registering recharge when the water levels in the hydrographs are not rising, while 1D model registers recharge in continuum.

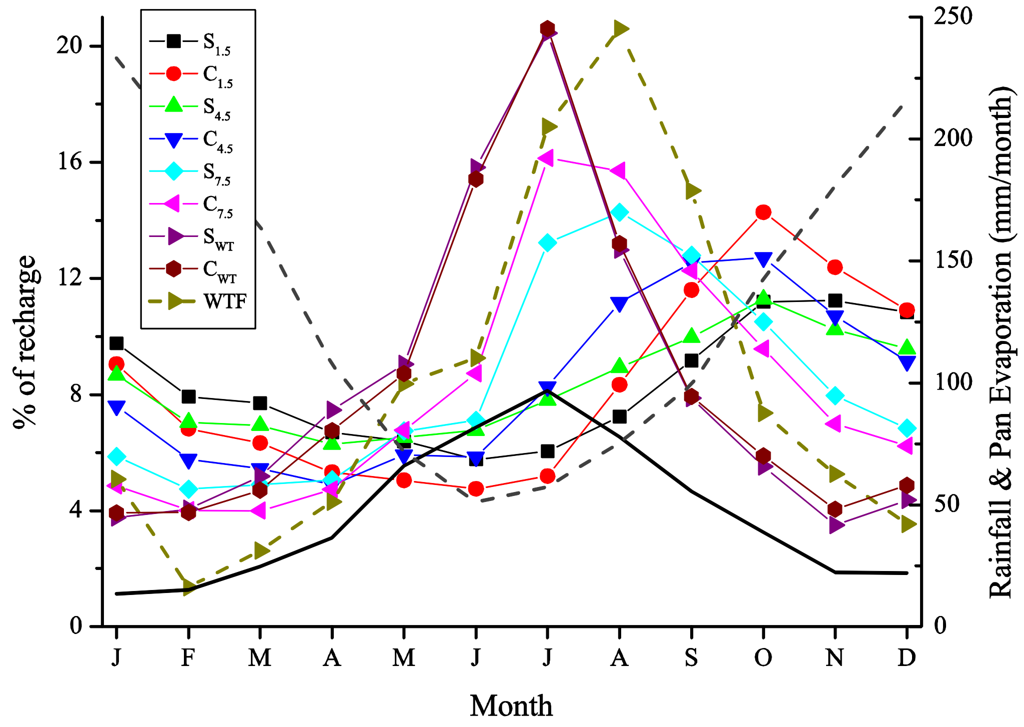

Figure 4.5 – Percentage of monthly average recharge rates for model- and WTF-based estimates. The solid black line represents average rainfall, and the dashed grey line represents average pan evaporation

Comparison between modelled and field-based timing of recharge on a well-to-well basis applying the NSC to monthly recharge rates is shown in Figure 4.6. Simulations $S_{7.5}$ and $C_{7.5}$ show the highest NSC values, reiterating that these simulations better represent the timing and seasonal contrasts in WTF recharge, as indicated in Figure 4.5. Particularly low NSC values are calculated for ULE126 and ULE194. Firstly, ULE126 is located very close to the aquifer boundary (Figure 4.1), and therefore water-table fluctuations in this region may be influenced more strongly by boundary conditions and thus not reflecting recharge predominantly. Secondly, WTF-based recharge estimates for ULE194 are based on a relatively small number of monthly water-table fluctuation measurements (168; see Figure 4.6 for number of monthly water-table fluctuations used for WTF-based recharge estimates in each well). This is generally much less than values for the other wells, and therefore limits the application of the NSC for inferring model-to-WTF-based agreement. Figure 4.6

also shows that NSC values for monthly recharge rates are generally below 0.5, indicating that the correlation between observed and modelled recharge rates on a monthly basis is not strong. This is, at least partially, a consequence of the differing representation of recharge using WTF and 1D modelling approaches, as aforementioned. This is emphasised more clearly in Figure 4.7, whereby WTF and modelled cumulative normalised monthly recharge rates are compared. Whilst the shape of the recharge curve is smooth and the recharge increment is continuous for the modelled results, it shows a step-wise pattern for WTF results. NSC values obtained for the comparison between WTF and modelled recharge, and summed to annual time-steps, are considerably higher than the NSC values for the comparison of monthly totals (typically between 0.25 and 0.75; not shown), and differences between different simulations are greatly attenuated. This shows that by integrating to annual time-steps, any issues with the monthly timing of recharge are largely neglected. An implication of this is that if annual recharge rates are sufficient for groundwater management purposes, input to groundwater models, etc., the recharge modelling using different scenarios of preferential flow representation become less important. If, however, modelled recharge rates at a finer temporal resolution are needed, which is arguably the case of USB (Werner et al., 2011), different scenarios of preferential flow infiltration should to be considered.

Justification for the differences between WTF and modelled timing of recharge are at this stage speculative; however, this could be evaluated further using a groundwater flow model. Similarly, Ireson and Butler (2011) justified the mismatch between their simulated recharge and water-table responses with lateral movements in the saturated zone. However, their study differs significantly from the current study as they used hourly field-data available (climatic and water levels), the modelling was for a shorter period (5 years), and recharge was modelled at the point scale and not integrated up to the basin scale. In this study, using field-based timing of recharge provided the only means of comparison and thus selection between different recharge simulations. We advocate the application of such an approach (i.e., constraining model-based recharge estimates using field-based methods) for cases where knowledge of transient recharge is required (e.g. groundwater management purposes, input to groundwater models).

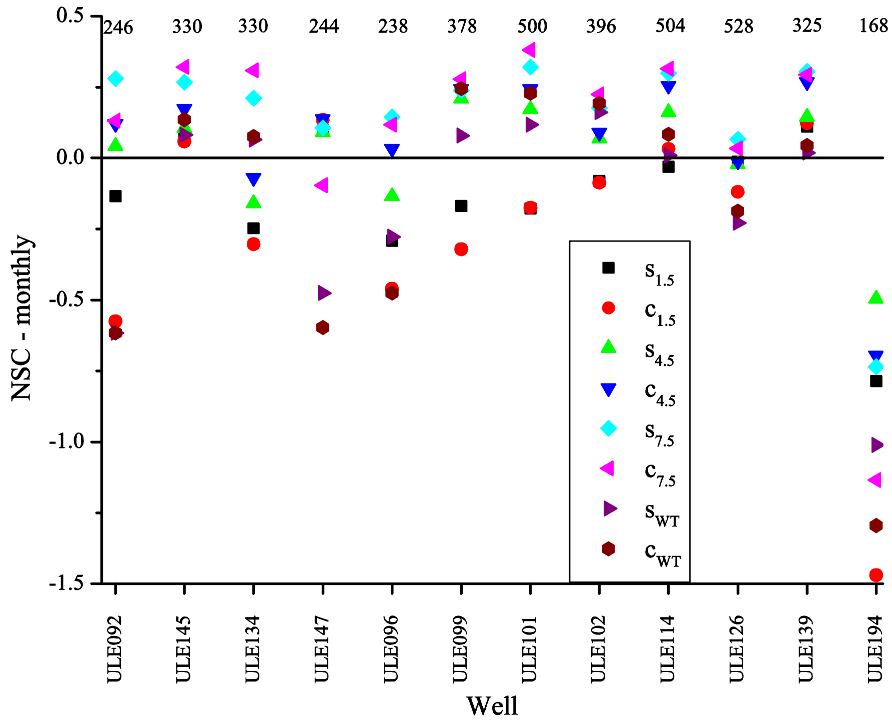

Figure 4.6 – NSC for monthly recharge rates for the different simulations and WTF, for each well. The values above the symbols are the number of WTF monthly recharge rates available for each well (the total number of simulated monthly recharge rates is 612)

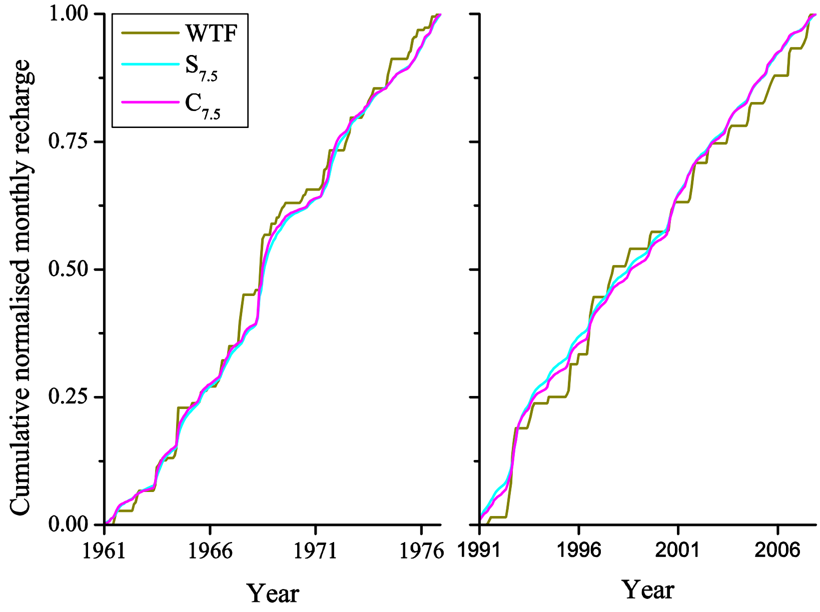

Figure 4.7 – WTF and modelled ($S_{7.5}$ and $C_{7.5}$) cumulative normalised monthly recharge rates for the periods 1961–1976 and 1991–2007 for ULE102 (no groundwater level data are available for the period 1977–1990 for this well)

Figure 4.8 shows the distribution of the ratio of annual-to-average recharge rates, versus annual rainfall. Only annual recharge rates for the periods 1961–2000 and 2003–2007 were used in Figure 4.8, which correspond to periods of available WTF data. Figure 4.8 indicates that both WTF and modelled annual recharge rates follow a pattern that is roughly consistent with the annual rainfall pattern, as expected. The relative WTF and simulated annual recharge rates are generally similar, which is an indication of consistency between the two approaches. The WTF and $C_{7.5}$ results show almost-coincident regression curves. Figure 4.8 also demonstrates that simulated annual recharge rates have a complicated, non-linear relationship with annual rainfall, which is suggested by relatively low r^2 values: WTF r^2=0.59, $S_{7.5}$ r^2=0.57 and $C_{7.5}$ r^2=0.72. For example, a rainfall of approximately 650 mm/year, which can result in a range of annual recharge rates (both modelled and WTF-observed) between approximately half and double the average recharge. These findings are consistent with conclusions of Ordens et al. (2012), who showed that although recharge generally follows the rainfall trend, annual rainfall and annual recharge are not linearly correlated. They suggested that factors other than annual rainfall influence annual recharge rates, such as temporal and spatial variability in evapotranspiration, antecedent unsaturated zone moisture conditions and the rainfall pattern (i.e. rainfall intensity, duration and location).

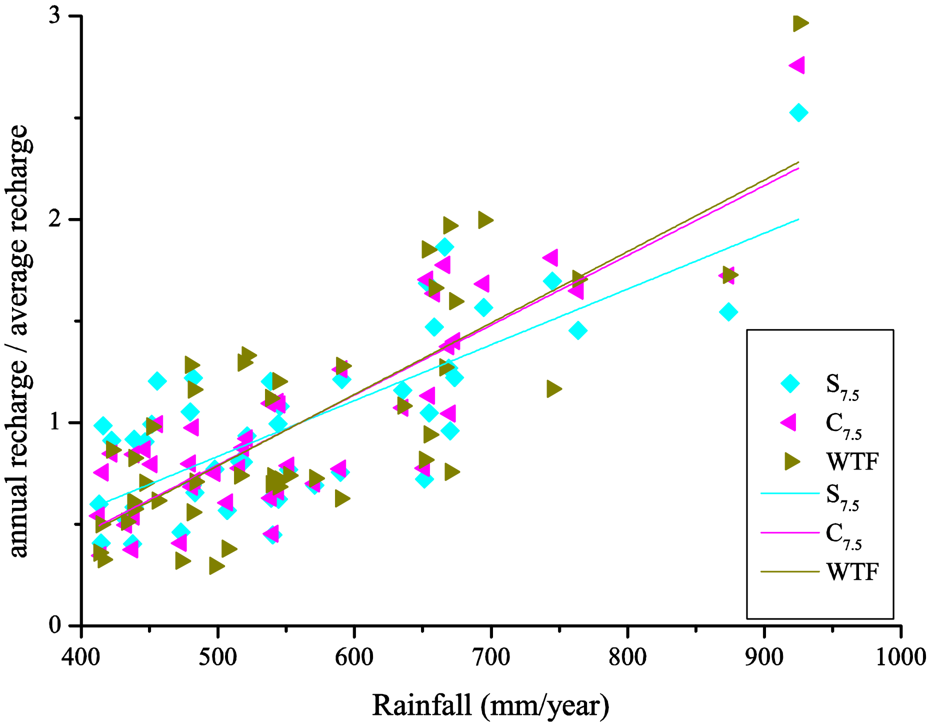

Figure 4.8 – Annual rainfall versus ratio of annual-to-average recharge rates from WTF (r^2=0.59), $S_{7.5}$ (r^2=0.57) and $C_{7.5}$ (r^2=0.72) for the period 1961-2007 (excluding 2001-2002 for lack of WTF data)

4.4.3 Implications for groundwater management

A description of the implications of modelled transient recharge rates for groundwater management purposes is given bellow, because it links fluxes (pumping and recharge rates) to changes in groundwater levels, and demonstrates the purpose behind applying transient recharge in the management of the aquifer. Werner et al (2011) showed that groundwater management practices in USB may have led to storage decline over the preceding decades. Water levels have approached mean sea level in places, raising the risk of seawater intrusion. At that time, USB management was based entirely on recharge estimates (Werner et al., 2011), using a time-constant rate of 140 mm/year. The results present here, and in previous studies by Werner et al. (2011) and Ordens et al. (2012), indicate that a lower average recharge value is more likely.

Figure 4.9 shows the average annual water levels for ULE101 (see well location on Figure 4.1), annual recharge from simulations $S_{7.5}$ and $C_{7.5}$, and groundwater extraction rates as reported in Werner et al. (2011). It shows a groundwater level decline since 1970. This decline was interrupted by a rise during the early 1980's that seems to be associated with years of high recharge. Pumping started in 1977, being of similar magnitude to recharge, and a significant drop in water levels (≈1 m) was observed for that period. Pumping was lower than recharge until 1994. From that year pumping has been approaching recharge rates, and in some cases exceeding those, which may be leading to the observed low groundwater levels. Groundwater levels appear to be reasonably stable for the period 1998–2007, although it may be because of the sea acting as a hydraulic barrier and not allowing the groundwater levels to drop further. It is not possible to distinguish the role of pumping and recharge on the groundwater levels in USB, which are also expected to be influenced by other factors such as inflows from adjacent basins, discharge to the sea and seawater intrusion, but the steepest declines in groundwater levels coincide with periods when pumping approaches recharge. A groundwater flow model of the basin would assist in properly exploring the relative importance of these different factors.

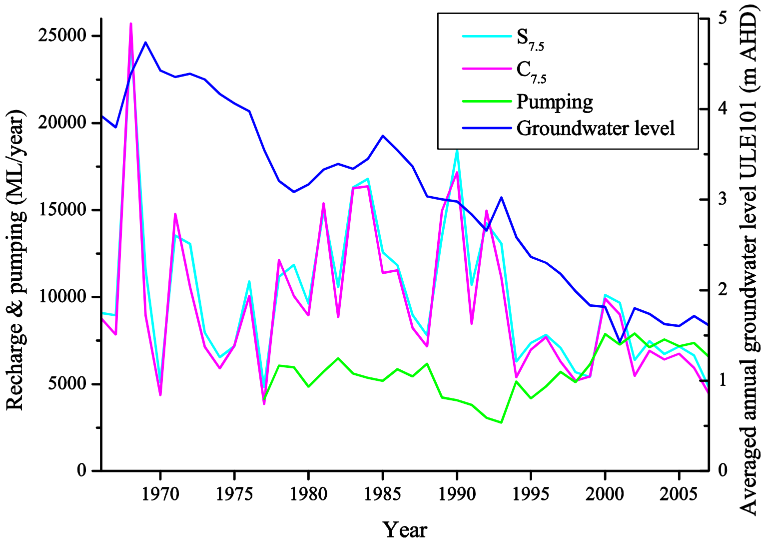

Figure 4.9 – Average annual groundwater levels for well ULE101, total basin recharge from simulations $S_{7.5}$ and $C_{7.5}$ (excluding sand dune area) and pumping rates

4.5 Conclusions

The combination of field-based methods and 1D modelling proved to be a successful approach to calculate temporally and spatially variable recharge rates to USB. This was a particularly effective methodology given that there was very limited unsaturated zone data to parameterize and validate the 1D model, and critically comparing field-based estimations and modelling results allowed for a validated recharge model. This is a useful approach that can be applied to other aquifers where little data exist to construct a recharge model, and where field-based estimates are available. Because 1D modelling of recharge can produce a very large range of possible values (e.g. Scanlon et al., 2002a), we advocate the use of field-based estimates to validate model predictions.

The simulations here considered result in spatially and temporally averaged recharge of 77 to 89 mm/year. Excluding the sand dune area, which is presently being considered (at least partially) out of the natural boundaries of USB (Knowling et al., in prep.), the modelled recharge rates are in the range of 61 to 74 mm/year. These values compare well with the field-based CMB estimates of 53 to 70 mm/year. The comparison of the field-based WTF with simulated timing of recharge, both on annual and monthly basis, showed that the best matches were found for the simulations $S_{7.5}$ and $C_{7.5}$. These simulations produced annual recharge rates that generally follow the annual rainfall trend in the basin, and are similar to the WTF trend. The best simulations of recharge to USB, $S_{7.5}$ and $C_{7.5}$, produced temporally averaged recharge rates of 89 and 84 mm/year respectively; excluding the sand dune, the recharge rates are 74 mm/year and 69 mm/year, respectively.

The comparison of CMB and simulated recharge rates on a well-to-well showed a mismatch between the two approaches. This is probably because the recharge areas of the wells do not coincide with the location of the wells. Although we attempted to determine recharge areas for the wells, it proved a complicated task mainly due to the aquifer being highly heterogeneous and given the different screen depths and intervals of the wells. Generally, higher CMB recharge rates are located down-gradient of areas of higher modelled recharge rates, which is an indication that recharge occurring at one point in the aquifer is measured as a groundwater chloride concentration at a point down-gradient (as expected). We interpret this as an evidence of consistency between the two approaches. Similarly, the comparison of field-based and modelled timing of recharge showed mismatches. This is not necessarily an inconsistency, but rather a result of (i) lack and poor quality of available field data, and (ii) water-table movements being affected by other processes than recharge (e.g. lateral movements). Further work could include using a groundwater flow model to define the recharge areas of the wells, which in turn would allow for a more precise comparison between the CMB and modelling results, and for the validation of the ability of the model to reproduce spatial variability of recharge. Likewise, simulated well hydrographs from a groundwater flow model would facilitate an improved comparison between the recharge timings from WTF and 1D recharge model, given that the areas over which WTFs apply are presently difficult to characterise, and because many of the water-level records are discontinuous and/or short-term, and wells are unevenly distributed across the basin. Most importantly, it would support a critical evaluation of the differences in recharge signal arising from the WTF and 1D modelling, as they represent different physical processes.

Acknowledgements: This work was co-funded by the National Centre for Groundwater Research and Training, a collaborative initiative of the Australian Research Council and the National Water Commission. The author Carlos M. Ordens would like to thank Fundação Ciência e Tecnologia (FCT), from the Portuguese Government, for funding his PhD grant (SFRH/BD/43625/2008). The author Matthew J. Knowling would like to thank the support by Australian Postgraduate Awards and Goyder Institute for Water Research PhD Supplements. The authors wish to thank the following people for helpful discussions in regards to this paper: James McCallum, Andrew Miller, Brooke Swaffer and Craig Simmons.

5 Applying simulated recharge to groundwater management of Uley South Basin

Note: the candidate is the third author of the journal paper (Werner et al., 2011) that forms the basis for this chapter. Therefore, this chapter was modified relative to the published paper, and only parts of the publication referenced in Chapter 1 (Introduction) are reproduced here to reflect the contributions of the candidate. The candidate was the main contributor for the "Case Study" section in the paper, which corresponds to the section "5.3 Study Area" in this chapter. Other sections are included for the sake of clarity of the over-arching problem that is presented here. For the full content of this paper, please see Werner et al. (2011).

Abstract

The control of groundwater abstraction from coastal aquifers is typically aimed at minimizing the risk of seawater intrusion, excessive storage depletion and adverse impacts on groundwater-dependent ecosystems. Published approaches to the operational management of groundwater abstraction from regulated coastal aquifers comprise elements of "trigger-level management" and "flux-based management". Trigger-level management relies on measured groundwater levels, groundwater salinities and/or ecosystem health indicators, which are compared to objective values (trigger levels), thereby invoking management responses (e.g. pumping cut-backs). Flux-based management apportions groundwater abstraction rates based on estimates of aquifer recharge and discharge (including environmental water requirements). This paper offers a critical evaluation of coastal aquifer management paradigms using published coastal aquifer case studies combined with a simple evaluation of the Uley South coastal aquifer, South Australia. There is evidence that trigger-level management offers advantages over flux-based approaches through the evaluation of real-time resource conditions and trends, allowing for management responses aimed at protecting against water quality deterioration and excessive storage depletion. However, flux-based approaches are critical for planning purposes, and are required to predict aquifer responses to climatic and pumping stresses. A simplified modelling

analysis of the Uley South basin responses to different management strategies demonstrates the advantages of considering a hybrid management approach that includes both trigger-level and flux-based controls. It is recommended that where possible, trigger-level and flux-based approaches be adopted conjunctively to minimize the risk of coastal groundwater degradation and to underpin strategies for future aquifer management and well-field operation.

5.1 Introduction

Threats to the freshwater resources of coastal aquifers include pumping-induced storage depletion, climate change impacts including sea-level rise, and pollution from urbanization. These potentially induce groundwater contamination via a range of possible sources and mechanisms (Jones et al., 1999; Werner and Gallagher, 2006; Alberti et al., 2009). Coastal aquifer management is commonly aimed at controlling seawater intrusion (i.e. the landward movement of the freshwater-saltwater interface), which is widely acknowledged as a significant threat to the availability of fresh groundwater resources globally (Post, 2005). Groundwater discharge to the sea and hydraulic heads control the inland extent of the interface. Consequently, estimates of the coastal aquifer water balance (i.e. recharge, discharge and changes in storage) are usually required, in combination with measurements and interpretations of groundwater levels, salinities and the position of the freshwater-saltwater interface, to guide management decision-making (FAO, 1997).

Seawater intrusion is typically a slow phenomenon (van Dam, 1999; Watson et al., 2010), and rapid interface movements are only occasionally observed (e.g. Melloul and Zeitoun, 1999). As such, short-term and localised aquifer behaviour are often neglected in management approaches (including well-field operational strategies), which rather tend to focus on long-term trends in pumping, recharge and freshwater-saltwater interface movements. However, contamination of fresh groundwater occurs at salinity levels of only 4% of seawater, and well contamination via saltwater up-coning can occur abruptly and is thought to be widespread across areas where pumping occurs above the freshwater-saltwater interface (Maimone et al., 2003). Some coastal aquifers, such as the freshwater lenses of small islands, are especially vulnerable to over-extraction, rapid salinisation through up-coning, and potential future climate change impacts (White and Falkland, 2010), and as such require more careful monitoring and regulatory control. The remediation of seawater intrusion is extremely challenging and resource intensive, and there are many instances of seawater-impacted coastal aquifers that cannot be restored to viable freshwater conditions (Custodio, 1987b; Maimone et al., 2003).

Coastal aquifer management requires adequate knowledge and assessment of seawater intrusion and the associated hydrogeological processes, including the complex nature of density-dependent solute transport in heterogeneous systems (Kashef, 1971; Post, 2005). There are many examples of seawater intrusion assessment using the predictions of numerical models, for the purposes of devising management controls (e.g. Gingerich and Voss, 2005). However, Sanford and Pope (2010) suggest that current methods are limited in their capability to predict individual well salinisation occurring from seawater intrusion at the regional-scale, and therefore managers of coastal aquifers are probably best informed by field-based measurements for guidance on the real-time status of coastal systems and related threats to pumping infrastructure. Given these factors, there is presently a need to review current management practices adopted in devising operational constraints on coastal groundwater systems, with a particular focus on strategies that identify and manage storage depletion and the associated seawater intrusion threat, such as the monitor-and-react approaches suggested by Evans et al. (2004) for inland aquifers.

In Werner et al. (2011) we focused on pumping operational and allocation strategies as one mode of coastal aquifer management amongst several others, e.g. physical barriers, hydraulic barriers, enhanced recharge. Only regulated coastal systems were considered. Two juxtaposed approaches to the management of coastal aquifers were evaluated, namely (1) flux-based management (FBM) and (2) trigger-level management (TLM). The former requires estimation of aquifer fluxes and is linked to concepts of sustainable yield (e.g. Bredehoeft, 2002; Alley and Leake, 2004). FBM is the traditional approach of many developed countries (e.g. NGC, 2004;

Lincoln Environmental, 2000). Conversely, TLM incorporates methods that adopt a somewhat continuous monitoring-management regime, using frequently monitored field conditions leading to adjustments in allowable groundwater extraction. Werner et al. (2011) offers a comprehensive review of coastal aquifer management approaches.

The primary objective of this study is to evaluate FBM and TLM approaches by summarising published examples of coastal aquifer management and through a simplified analysis of the Uley South Basin (USB) – a coastal aquifer situated on the Southern Eyre Peninsula, South Australia that serves as the principal water supply for the residents of that region. The USB case study was used to examine the performance of the aquifer (approximated from a simple modelling approach) under various management controls, and the findings were used in combination with "lessons learnt" from other published accounts of FBM and TLM to offer a reflective assessment of management practices for regulated coastal groundwater systems. The USB case study allowed for a direct comparison of FBM and TLM approaches that has otherwise not been undertaken. The results provided initial insight into over-arching ideologies for designing water allocation rules and ongoing management principles, and are likely to have relevance for regulators of other coastal groundwater systems.

5.2 Defining FBM and TLM

FBM is typically manifested as constraints on groundwater extraction volumes, which are imposed up-front (e.g. annual extraction volumes allocated to water users on a once-a-year basis), and these are meant to pre-empt and circumvent degradation of the groundwater resources and associated environments (e.g. Don et al., 2006; Heyns, 2008). Water allocations are usually assigned following an assessment of the system response to different aquifer stresses, often using computer models. FBM applied to coastal aquifers is contingent on the estimation (and prediction) of both aquifer recharge and submarine groundwater discharge; these are challenging to determine in the vast majority of circumstances (e.g. Scanlon et al., 2002a; Taniguchi et al., 2006).

FBM approaches accord several operational advantages, including forward planning of groundwater use by both water resource managers and end-users. Importantly, an increasingly robust understanding of system dynamics often results from periodic analyses of water balance fluxes over extended periods of analysis (Bredehoeft, 2002). However, there is considerable uncertainty regarding estimation of recharge volume, as required under FBM strategies, and also localised pumping effects are neglected (Evans et al., 2004). There are many cases of coastal aquifer degradation that have been linked to traditional FBM pumping allocation, illustrating the shortcomings of a non-adaptive management approach; for example the Choushui River alluvial, Taiwan (Liu 2004), north-eastern Korinthia aquifers, Greece (Voudouris 2006), amongst others.

TLM relies on measured groundwater levels, groundwater salinities and/or ecosystem health indicators, which are compared to objective values (trigger levels) thereby providing resource managers with a real-time basis for controlling groundwater extraction (e.g. Evans et al., 2004; Werner, 2010). For example, pumping cut-backs may be imposed in response to low or reducing storage levels or increasing groundwater salinities. TLM is often referred to as “groundwater level response management” (e.g. Evans et al., 2004; Bekesi et al., 2009), where storage levels alone are adopted in managing groundwater extraction. TLM is a more holistic aquifer monitor-manage approach for coastal aquifers compared to groundwater level response management, because TLM allows for the inclusion of salinity values and trends and ecosystem health indicators in assessing aquifer condition.

Being adaptive in nature, TLM does not require a priori recharge estimation. Further, TLM accommodates temporal and spatial variability in both groundwater pumping and rainfall recharge (Evans et al., 2004). TLM strategies facilitate the micro-management of localised pumping effects, which allows for the protection of key groundwater dependent ecosystems and high-priority production wells (e.g. urban water supplies) by avoiding excessive water table decline at specific points in the basin (NGC, 2004). However, the TLM approach requires determination of both trigger levels and management response actions (e.g. pumping cut-back specifications), and effective management is contingent on selecting parameters which are appropriate for monitoring the full spectrum of risks faced. TLM applications need to be flexible and evolutionary, such that triggers can be fine-tuned based on historical performance of the TLM regime.

We intentionally define the two approaches as contrasting management paradigms to highlight the strengths and weaknesses of each, and also we consider combined or hybrid approaches that adopt elements of both FBM and TLM (e.g. Liu et al., 2006). Under the definitions described here, the two approaches differ in that FBM is considered as a pre-emptive management approach, whereas TLM is reactive and based on current conditions. Also, FBM relies heavily on a reasonably accurate assessment of the aquifer water balance, as per the safe yield study of Voudouris (2006), while TLM is underpinned by aquifer heads, salinities and other field-based measurables. Evans et al. (2004) further distinguishes the two approaches based on spatial scale, with FBM being applied at the basin scale and TLM applicable at the sub-catchment scale thereby allowing for variations in local conditions and hydrologic boundaries in applying constraints on groundwater access. The application of either FBM or TLM for mitigating (or avoiding) seawater intrusion requires appraisal of groundwater abstraction (preferably through metering) to ensure compliance with pumping restrictions, which are set to levels considered necessary to protect fresh groundwater resources from degradation (FAO, 1997). Determining allowable pumping rates under both FBM and TLM approaches is challenging from a hydrological perspective, but also from the perspective of the relevant socio-economic aspects of water supply management. In some cases, water restrictions may be set as limitations on the timing of groundwater pumping, rather than volumetric measures, in cases where metering is inadequate (NRMSC, 2002). Both FBM and TLM require predictive analysis of the coastal aquifer in order to establish allocation limits based on recharge (in the case of FBM) or trigger levels and pumping cut-backs that protect the resource during periods of excessive stress (in the case of TLM).

Predictive computer modelling simulations of hypothetical TLM strategies (e.g. using optimisation methods) may be used to assign water allocations, and as such, proposed/hypothetical TLM approaches may be considered in assigning FBM pumping restrictions (e.g. Reichard and Johnson, 2005). Whether or not TLM is subsequently enacted as an operational control and the degree of effectiveness of TLM approaches are rarely reported in the literature. In the extreme situation of an unmanaged coastal aquifer, a natural form of TLM occurs whereby groundwater extraction is limited by hydrogeological and chemical constraints, e.g. excessive increases in salinity or falling water-table conditions that inhibit groundwater use. In these cases, environmental constraints impact on both the use and access of groundwater, thereby restricting pumping rates and volumes due to the uncontrolled degradation of the resource and limits to the use of brackish/saline groundwater. It is common under these conditions to find social inequities in the accessibility to groundwater resources within individual communities due to the rising cost of constructing deeper wells to access falling groundwater levels (e.g. van Steenbergen, 2006).

5.3 Study area

The USB case study serves to provide a comparison of FBM and TLM (and hybrid FBM-TLM approaches) applied to the regulation of groundwater pumping in a coastal aquifer setting (Figure 5.1). It is not possible to undertake field-based testing of proposed management strategies for the USB, and so a simple predictive model is adopted to evaluate the aquifer responses to different management approaches. The model is based on a parsimonious water-balance method that includes rudimentary predictions of the landward incursions and excursions of the toe of the saltwater-freshwater interface, and also considers the storage of seawater in the aquifer as part of the water balance.

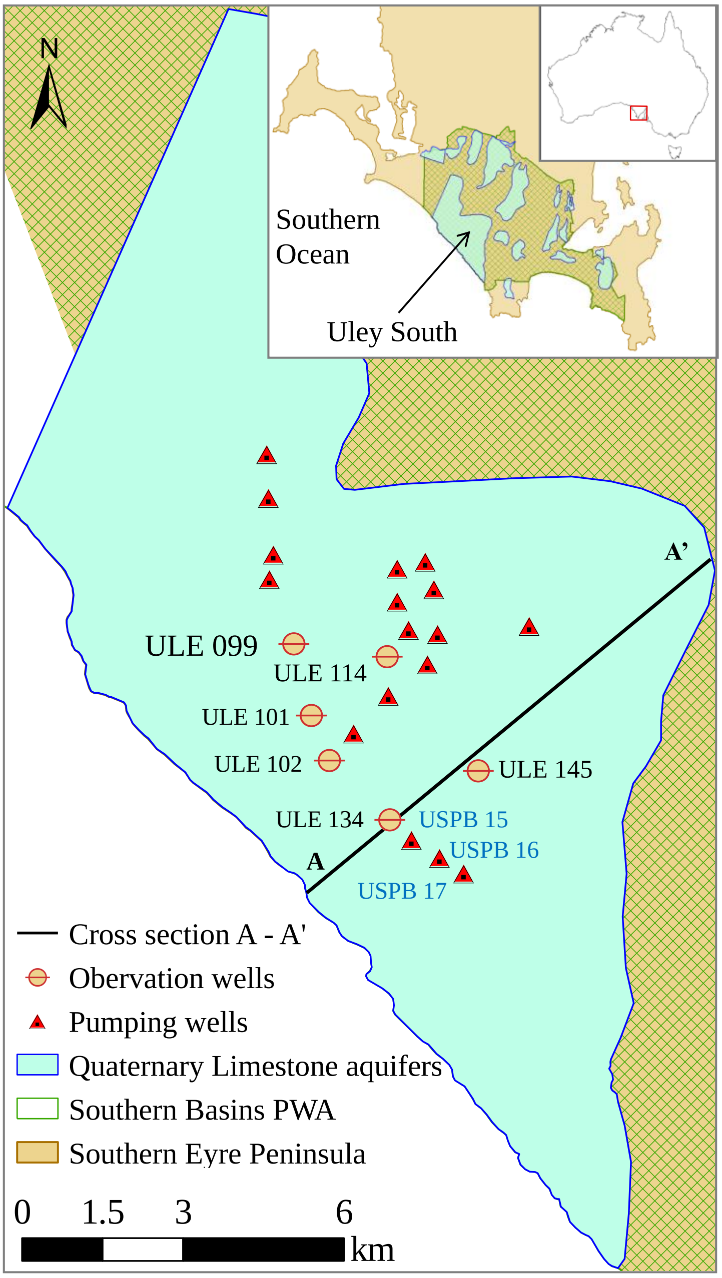

Figure 5.1 – Map of the Uley South Quaternary sandy limestone extent, showing the location of production wells and observation wells used for model calibration. The large inset shows USB and other neighbouring basins within the Southern Basins Prescribed Wells Area (PWA)

The community of Eyre Peninsula (EP) have raised concerns regarding security of their public water supply (EPNRM 2006), which is highly dependent on USB groundwater abstraction, accounting for around 70% of the EP's total reticulated water needs (Zulfic et al. 2007). Persistent trends of groundwater level decline can be observed within several EP aquifers, including in the USB (Zulfic et al. 2007), and the threat of saltwater intrusion has been identified for aquifers adjoining the coast. A typical groundwater hydrograph for the basin is shown in Figure 5.2, which also illustrates monthly rainfall totals. Sustainable groundwater extraction from USB is a key priority for water management authorities. Presently, rights to extract groundwater from each of EP's Prescribed Wells Areas (PWAs) are "allocated" to users in terms of volume, in accordance with Water Allocation Plans (EPNRM, 2006); i.e. a FBM paradigm is adopted. Estimates of recharge are gazetted each November, and allocations are then announced for the following financial year (1st July to 30th June). Under the current FBM arrangement, 60% of USB rainfall recharge is reserved for the needs of groundwater-dependent ecosystems and 40% is available for allocation (EPNRM, 2006), although Harrington et al. (2006) report that there is considerable uncertainty in estimates of recharge volumes and sustainable yields used in assigning water allocations.

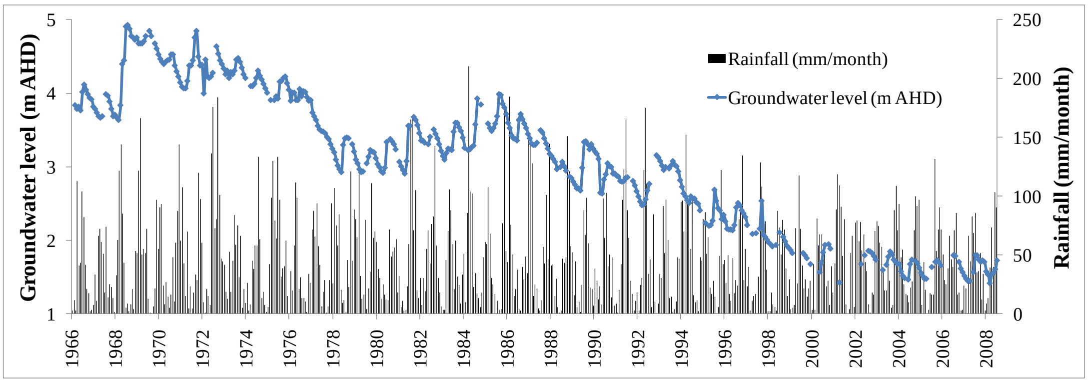

Figure 5.2 – Water level hydrograph from well ULE101 and monthly rainfall totals

The climate across the southern EP is semi-arid to sub-humid, with a predominant cold wet season occurring from May to October and a warmer dry season from November to April (Harrington et al. 2006). Rainfall is highly variable across the EP, ranging between 351 and 925 mm/annum (Harrington et al. 2006). The USB lies within a topographically enclosed catchment and is essentially uninhabited. The gently undulating plains of the central portion of the basin are bounded by coastal dunes (up to ~140 m AHD; Australian Height Datum (AHD) is approximately mean sea level) at the Southern Ocean boundary, and by low relief (~70-140 m AHD) ranges around the remaining catchment perimeter.

The hydrogeologic units present in the basin are, from top to bottom: (1) Quaternary sandy limestone aquifer (QL), which comprises unconsolidated or loosely aggregated aeolic sands with intercalations of calcrete bars and clay layers, showing evidence of both primary and secondary porosities; (2) Tertiary clay aquitard (TC), a clay layer that is known to be absent in some parts near the coast; (3) Tertiary sand aquifer (TS); and (4) Proterozoic volcano-sedimentary basement sequence. The maximum thicknesses of the QL, TC and TS sequences are 130 m, 25 m and 60 m, respectively; Figure 5.3 illustrates a cross-section through the central part of the basin. The groundwater flow direction is primarily northeast-to-southwest, as illustrated in Figure 5.4. This study is focused on the QL aquifer, as it is the subject of groundwater abstraction, although in some parts QL and TS can be hydraulically connected. Physiographic attributes of the study area are described in detail by Harrington et al. (2006).

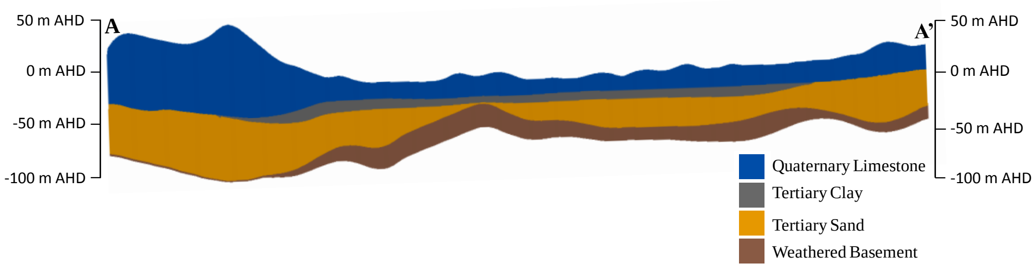

Figure 5.3 – Cross-section through A-A'. Cross-section alignment is shown on Figure 5.1

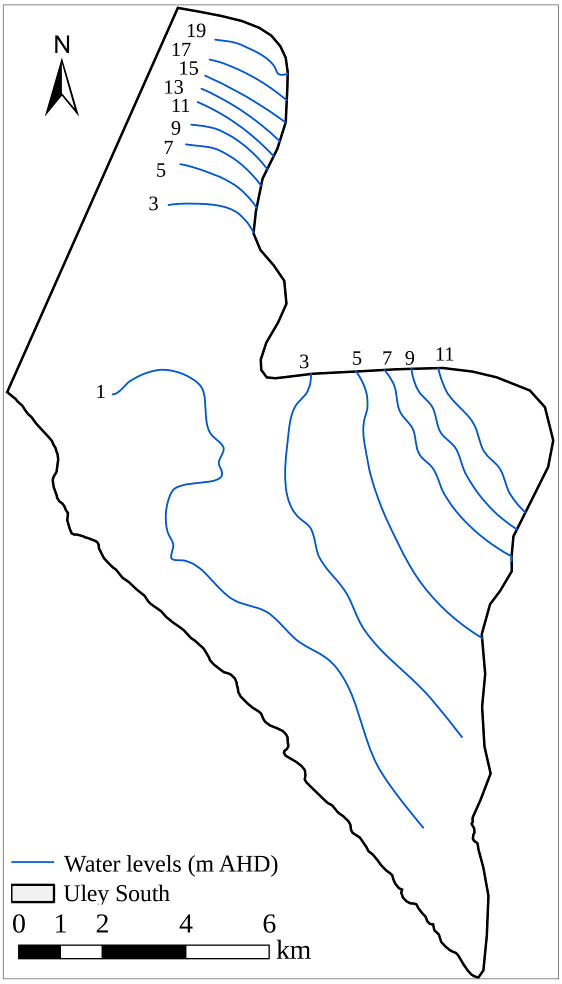

Figure 5.4 – USB groundwater levels (from 2007 measurements) in the QL aquifer

5.4 Methodology

A lumped-parameter, transient water balance model with a first-order approximation of seawater intrusion extent was used. The model's governing equation is based on the indirect or "residual" water balance approach (Scanlon et al., 2002a), where the rate of change in groundwater storage is calculated as the arithmetic difference between rates of groundwater inflow and outflow:

$$\Delta S = W_{RCH}A + Q_{UE} + Q_{UW} + Q_{CB} - (Q_{PUMP} + Q_{SEA}) + Q_{SWI} \qquad (5.1)$$

where ΔS is the rate of change in groundwater storage [L^3/T], W_{RCH} is the spatially averaged net rainfall recharge accounting for groundwater evapotranspiration [L/T],

A is the planar area of USB [L^2], Q_{UE}, Q_{UW} and Q_{CB} are groundwater inflows [L^3/T] from surrounding basins of Uley East (UE), Uley Wanilla (UW) and Coffin Bay (CB) lenses, respectively, Q_{PUMP} is aggregated groundwater pumping [L^3/T], and Q_{SEA} is groundwater discharge to the sea [L^3/T]. Q_{SWI} represents the rate of change in the volume of seawater stored in the aquifer [L^3/T] (positive for an increase in seawater volume), as defined by the quasi-steady-state position of the freshwater-seawater interface.

The position of the interface was assumed to follow the theoretical steady-state position (equations are given below) as determined from average monthly aquifer recharge and hydraulic head conditions, thereby producing a transient simulation of interface movement using the simplifying assumption that the interface retains a steady-state distribution. The addition of seawater volumes to the water balance and estimation of the position of the interface toe are improvements over hitherto modelling attempts of the USB, e.g. by Harrington et al. (2006). A monthly time-step is adopted. Changes in groundwater storage are converted to water-table fluctuations using: $\Delta h = \Delta S/(AS_y)$, where Δh is the simulated monthly change in groundwater level [L/T], and S_y is specific yield [–]. These are used to produce a predicted groundwater hydrograph, which is then compared to historical water-table trends in USB.

Time-series of recharge inputs to the water balance model were area-averaged values from the LEACHMG simulations (chapter 4). Groundwater inflows to USB from adjacent lenses (Q_{UE}, Q_{UW} and Q_{CB}) were estimated through Darcy's Law and using typical aquifer parameters from previous studies (e.g. Harrington et al., 2006). Groundwater pumping data were sourced from metered records maintained by South Australian Water Corporation, who is the sole groundwater user of the basin. Groundwater discharge to the sea was calculated considering the occurrence of the freshwater-seawater interface (assuming that quasi-steady-state conditions prevail), via (Custodio, 1987a):

$$Q_{SEA} = L_C \frac{h^2 K(1+\alpha) + W_{RCH} x^2}{2x} \tag{5.2}$$

where L_C is the length of USB coastline through which groundwater discharge occurs [L], h is the steady-state groundwater head above sea level [L] and at some distance x from the coastline [L], K is hydraulic conductivity [L/T], and α is the density ratio $\rho f/(\rho s - \rho f)$ [-], where ρf and ρs are freshwater and seawater densities [M/L^3], respectively. The time-series of h used to estimate interface movements corresponds to the groundwater hydrograph obtained from the water balance calculations described above.

The volume of seawater in the aquifer was approximated from the theoretical steady-state freshwater-seawater interface distribution, given as the elevation $z(x)$ of the interface above the basement [L]. The equation for $z(x)$ was derived by manipulating (5.2) and considering that groundwater flow in the coastal fringe qi [L^2/T] at some distance x_i [L] from the coastline equates to $q_i = Q_{SEA}/L_C - W_{RCH}x_i$ (assuming equilibrium flow conditions). $z(x)$ is therefore given as:

$$z(x) = B - \alpha\sqrt{\frac{2q_i x + W_{RCH}\,x(2x_i - x)}{K(1+\alpha)}} \qquad (5.3)$$

Equation (5.3) was numerically integrated to estimate seawater volumes within the aquifer, and these allowed for the approximation of Q_{SWI} [L^3/T]. An important hydrogeological variable in both the management and modelling frameworks is the location of the freshwater-seawater interface toe, which is defined as the inland distance x_T [L] at which the interface intercepts the basement. The toe is used as a surrogate for quantifying seawater intrusion changes, and is calculated under steady-state conditions as (Custodio, 1987a):

$$x_T = \frac{Q_{SEA}}{L_C W_{RCH}} - \sqrt{\frac{Q_{SEA}^2}{L_C^2 W_{RCH}^2} - \frac{K(1+\alpha)B^2}{W_{RCH}\alpha^2}} \qquad (5.4)$$

The above theoretical framework allowed for the prediction of USB storage levels and representative hydraulic heads at some distance from the coastline. Model predictions were compared to the hydrographs from six observation wells by running the model six times and modifying x_i within the model construct to accord with each of the six well positions. The calibration match (i.e. observation versus simulation) was assessed by evaluating the mean calibration statistics across all six observation wells; an approach considered adequate for an undistributed model. K and S_y were varied during automated calibration (using the Microsoft® Excel Solver Package; e.g. Ayenew & Gebreegziabher, 2006), subject to parameter constraints based on field evidence from previous studies (e.g. Harrington et al. 2006). Model performance was assessed using a combination of the root-mean-square error ($RMSE$) and the Nash-Sutcliffe Coefficient of Efficiency (E_f). $RMSE$ and E_f in combination account for both relative and absolute errors (Legates and McCabe, 1999; Middlemis et al., 2000).

The model was used to test different groundwater allocation scenarios, including: (i) unrestricted pumping – water supply and demand are equal; (ii) FBM – pumping is restricted when demand exceeds the allocation of 40% of 10-year average recharge (i.e. a USB "business-as-usual" scenario as defined by the Water Allocation Plan; EPNRM, 2006); (iii) TLM – pumping decreases or ceases when groundwater levels fall below the seawater intrusion trigger level/s (described below); (iv) hybrid FBM-TLM – pumping is restricted when demand exceeds allocation or pumping decreases or ceases when groundwater levels fall below the seawater intrusion trigger level/s; and (v) adaptive FBM or FBM-TLM - pumping is restricted when demand exceeds allocation, but pumping can be higher or lower than present allocation based on the performance of the aquifer (in terms of groundwater levels) in the previous year. Water demand (and therefore supply) follows a seasonal pattern as per trends in metered pumping. Aquifer performance in the previous year modifies annual allocation in adaptive FBM scenarios. In adaptive scenarios involving TLM approaches, the aquifer water levels are used to allow higher pumping (for periods of high water tables) and/or to enforce restrictions when water levels fall below trigger levels. TLM and FBM pumping cut-backs are invoked at a monthly time-step in the model.

Seawater intrusion trigger levels were set to reduce the risk of well-field salinisation. The trigger level used to invoke stoppage of pumping was taken as the hydraulic head that is required to maintain the condition x_T < 2100 m; this being the approximate inland distance to productions wells USPB15, USPB16 and USPB17 (assuming an aquifer thickness of 50 m). A trigger level of 1.29 m AHD was obtained using the theoretical framework described above and calibrated parameters. Additionally, trigger levels of 1.35 m AHD, 1.48 m AHD and 1.80 m AHD, producing x_T equal to 200 m, 500 m and 1000 m (from near-coastal production wells), respectively were used to modify allowable pumping rates. In a real-world situation, these trigger levels would be adapted based on field observations of interface extent under varying hydrologic conditions.

Three parameters were used to evaluate the relative efficacy of groundwater allocation scenarios: (i) predicted annual rate of groundwater pumping (ML/year), which was used as a measure of water availability; (ii) number of months (sojourns) when the groundwater level fell below the trigger level of 1.29 m AHD; and (iii) number of months when no groundwater pumping was allowed. An optimal management regime maximises water availability while minimising both the risk of seawater intrusion (assumed linked to the number of sojourns) and the number of months when groundwater pumping is not allowed. Predictive simulations used historical recharge patterns as input to the model, and therefore the basin performance is tested assuming that future rainfall will follow previous trends. The description of the different scenarios is given in Table 5.1

Table 5.1 – Description of the different management scenarios applied to USB

Scenario	Description
No restrictions	Pumping equals water demand
FBM 1	Pumping restricted when demand > 40% of 140 mm/year recharge
FBM 2	Pumping restricted when demand > 40% of 10-year average recharge
TLM 1	Pumping ceases when water levels < 1.29 m AHD
TLM 2	Pumping ceases when water levels < 1.35 m AHD
TLM 3	Pumping ceases when water levels < 1.48 m AHD
TLM 4	As per TLM 1, and pumping reduced (by 50%) when water levels < 1.35 m AHD
TLM 5	As per TLM 4, and pumping reduced (by 25%) when water levels < 1.48 m AHD
TLM 6	As per TLM 5, and pumping increases (by 10%) when water levels > 1.8 m AHD
Hybrid 1	Combination of FBM 1 and TLM 1
Hybrid 2	Combination of FBM 2 and TLM 1
Hybrid 3	Combination of FBM 2 and TLM 5
Hybrid 4	Same as Hybrid 3 except allocation is based on 140 mm/year recharge
Hybrid 5	Combination of FBM 2 and TLM 6
Adaptive FBM	Same as FBM 2, except allocation increased based on previous year performance, via: +10% if sojourns=0, no change if sojourns=1, -10% if sojourns>1
Adaptive Hybrid	Combination of Adaptive FBM and TLM 6

5.5 Results and Discussion

Calibration of the model produced a value for *K* at the coastal boundary between 160 m/d and 300 m/d, and *Sy* of 0.14 to 0.20. The parameter ranges given here represent the minimum and maximum values obtained by individually calibrating six USB models to six different observation wells, as mentioned previously. Calibration performance measures (averaged across the six different USB models) indicate a favourable "goodness-of-fit" between observed and predicted water levels ($RMSE$ = 0.24 m and E_f = 0.91), which is supported by the calibration curves depicted in Figure 5.5.

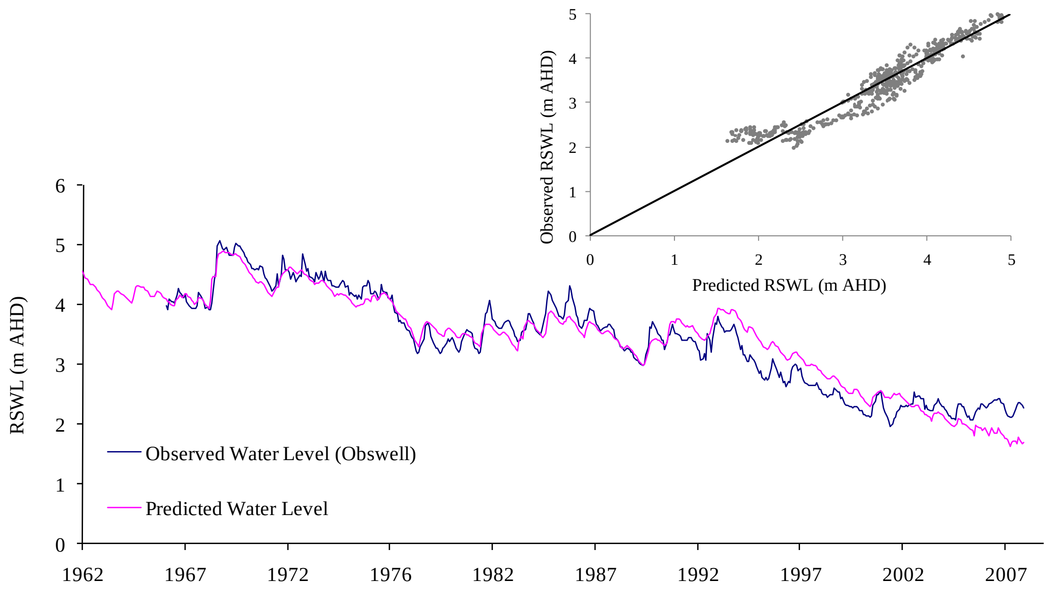

Figure 5.5 – Simulated and observed water-table dynamics and a scatter plot of the calibration match

The simulated USB water balance in terms of cumulative flows during the period 1967-2007 is illustrated in Figure 5.6. Rainfall recharge is the main input to the USB, estimated as 14 GL/year (1967-2007), which is lower than the value of 23 GL/year used by Zulfic et al. (2007) in their steady-state model of USB. Simulated inflows from the surrounding basins of UE, UW and CB are relatively minor, consistent with the findings of Harrington et al (2006).

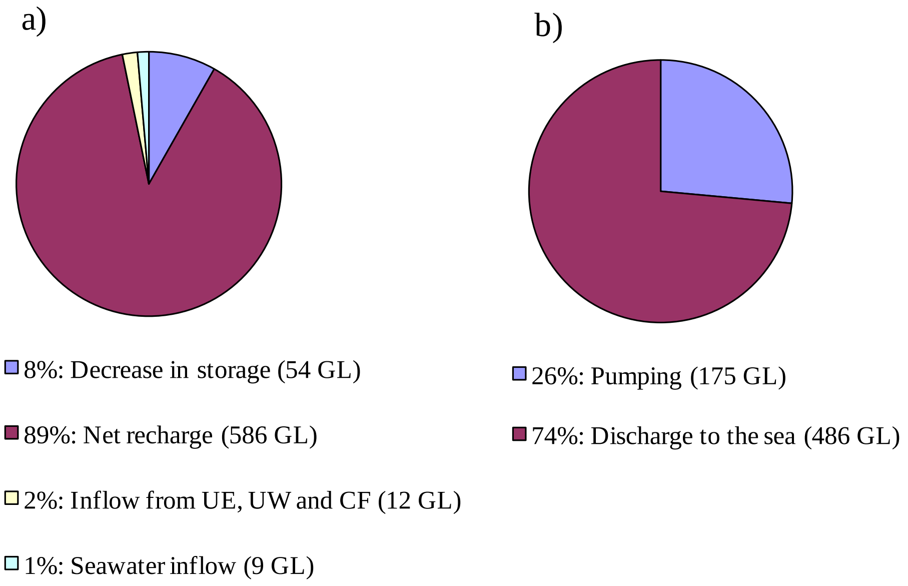

Figure 5.6 – USB water balance (1967-2007) (a) cumulative inflow (includes a 54 GL decrease in storage); (b) cumulative outflow

The modelling results indicate that the toe location of the saltwater-freshwater interface may have migrated inland in the USB, in the order of 170 m (median of the six calibration wells) from its position in the late 1960s. Although the current model results show that the cumulative flux of seawater is small relative to cumulative recharge, the predicted increase in seawater volume in the aquifer (9 GL) is significant relative to the change in groundwater storage of 54 GL – i.e. the reduction in stored freshwater in the QL aquifer is 63 GL. Clearly, the influx of seawater into the aquifer has offset freshwater storage reductions; neglecting this phenomenon results in an over-estimate of the available freshwater in the USB aquifers, and therefore has important implications for future water resource allocation strategies. This point highlights the need to instigate strategic monitoring of the freshwater-seawater interface and also adaptation of management practices to incorporate TLM to protect against future well salinisation and excessive groundwater storage losses.

The performances of the different management methods are shown in Figure 5.7. Note that the use of Ordens et al. (2010) estimates of recharge (i.e. averaging

107 mm/year) rather than the recharge estimate presently considered (EPNRM, 2006) (i.e. averaging 140 mm/year) means that FBM approaches adopted in the model impose more severe pumping restrictions than have been historically enforced.

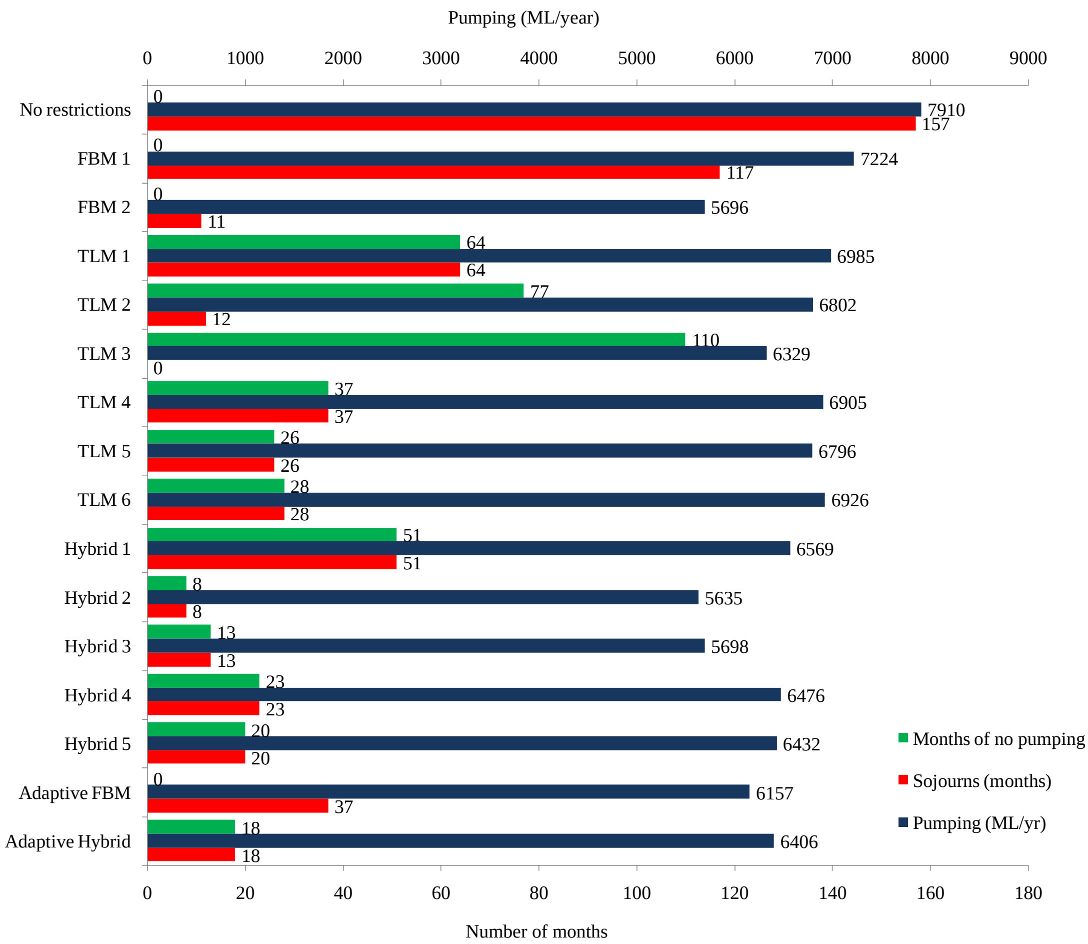

Figure 5.7 – Comparison of the results from modelling scenarios

The results given in Figure 5.7 indicate that implementing FBM 1 (i.e. a surrogate for the current management regime) produces significant reductions in both sojourns (i.e. from 157 to 117) and allowable pumping (i.e. from 7910 ML/year to 7224 ML/year), relative to the “no restrictions” scenario. FBM 2 adopts a reduced rate of recharge based on recent estimates by Ordens et al. (2010), and this produces further pumping cut-backs (i.e. to 5696 ML/year) and a major decrease in sojourns (i.e. to 11). These differences highlight that the successful application of FBM may be highly dependent on accurate estimates of recharge, which are often difficult to obtain (e.g. Scanlon et al. 2002a). Further, an inaccurate estimation of recharge led to an enhanced seawater intrusion threat (i.e. in the form of a large number of sojourns) under the FBM regime simulated for the study area.

The implementation of the TLM 1 scenario produced a large number of both sojourns and months where pumping was prohibited (i.e. 64). Note that sojourns and the number months of prohibited pumping are the same because restrictions occur at 1.29 m AHD, which coincides with the sojourn breach level. Despite the large number of zero-pumping months, TLM 1 achieved high water availability (6985 ML/year). Two other trigger levels were tested, namely scenarios TLM 2 and TLM 3 (the sojourn breach elevation of 1.29 m AHD was retained in these cases). Raising the trigger level (in TLM 2 and TLM 3) slightly reduced the allowable volume of pumping and significantly reduced the number of sojourns (see Figure 5.7); however substantial increases in the number of zero-pumping months were incurred, likely to unacceptable levels given that USB is used for urban water supply.

Tiered approaches to the TLM method (i.e. TLM 4, TLM 5, TLM 6) were tested in an effort to reduce the number of zero-pumping months. Compared to the single trigger level TLM approaches, the tiered TLM regimes produced a significant drop in sojourns and months of no pumping, and only a slight reduction in allowable pumping (see Figure 5.7). Note that TLM 6 adds the possibility of pumping 10% more than demand when the groundwater level is high, i.e. anticipating that the water supply infrastructure is able to capitalise on this “over-extraction” during high water-table periods.

The evaluation of both FBM and TLM led to the assessment of six hybrid FBM-TLM approaches, these being combinations of the previous scenarios. Ultimately, hybrid approaches invoke stricter controls on extraction, and therefore these led to fewer sojourns and months without pumping, and generally allowed lower volumetric groundwater extraction (see Figure 5.7). The different recharge rates (assumed in the hybrid FBM-TLM management approaches) again impacted on the aquifer performance outcomes, in terms of sojourns, water availability and number of zero-pumping months, although the error in recharge was alleviated by the addition of TLM, relative to the FBM-only strategies. The USB model results indicate that the potential to mismanage the aquifer (e.g. due to incorrect estimates of recharge) may be reduced with the addition of TLM to the FBM approach.

The previous FBM, TLM and hybrid FBM-TLM approaches neglect the capacity for water resource managers to adapt allocations based on aquifer performance. To address this, two adaptive scenario were assessed, and these involved allocation changes based on the number of sojourns occurring in the previous year (see Table 2). The adaptive FBM scenario performed relatively poorly compared to hybrid approaches, but outperformed the original FBM cases (FBM 1, FBM 2). The adaptive hybrid scenario performed similarly to the scenario Hybrid 5, and these were considered to be the optimal management strategies because they achieved high pumping and low sojourns and zero-pumping months (see Figure 5.7). Obviously, such factors as socioeconomics, water-supply security and environmental health implications need to be considered to properly evaluate these results in terms of the inherent trade-offs between allowable pumping and the risk of aquifer degradation.

Our contention is that a hybrid allocation strategy is considered the most pragmatic management approach because benefits inherent to tiered TLM (e.g. water resource protection) and adaptive FBM management (e.g. a continually enhanced understanding of system behaviour) are encapsulated in a single approach. Further, the benefits of each approach are complementary, because an enhanced understanding of system behaviour aids in defining (and successively redefining) trigger levels and FBM pumping percentages. The modelling used in this study demonstrates the application of a parsimonious management tool to support decision making regarding water availability and risk of aquifer degradation, inferred by the threat of seawater intrusion in the present case. The water levels adopted as trigger levels in this study may be supplemented by such field observations as well salinity values (i.e. observation wells and production wells), and also water level and salinity trends can be considered rather than absolute values to better capture the nature of changes in aquifer condition.

5.6 Conclusions

We have tested various operational strategies for managing extraction from the USB coastal aquifer. A non-adaptive pumping management regime (FBM-based) has been adopted previously for the USB, and the groundwater levels have fallen steadily across a 30-year period, approaching sea level in some places. A parsimonious water-balance model is adopted to explore different pumping management approaches in the USB. The results indicate that the addition of TLM to the FBM-based management of the system leads to (i) enhanced water availability manifested as higher allowable pumping volumes and fewer zero-pumping months; (ii) a reduction in the risk of aquifer degradation (i.e. reduced sojourns/risk of seawater intrusion) and protection against recharge estimate inaccuracies; and (iii) an enhanced understanding of basin functioning leading to adaptive management. The present study also demonstrates the importance of performance-based (in terms of aquifer health) adaptation in the allocation of pumping; adaptive scenarios provided optimal water supply and resource security outcomes.

The USB case study in combination with the review of previous studies demonstrates that rigid flux-based approaches to coastal aquifer management may not adequately protect the aquifer from degradation nor provide optimal extraction volumes. The lack of TLM measures in reported accounts of aquifer management needs to be addressed, although trigger-level constraints need to be implemented judiciously, likely using a tiered approach, to avoid both over-regulating the aquifer and unnecessarily reducing available extraction. Further consideration should be given to the application of water level and salinity trends as trigger levels (rather than just absolute values) in developing coastal aquifer management strategies. In addition, given that the risk of seawater intrusion is closely related to the discharge of the aquifer to the sea, this component of the water balance, which is often neglected, should be taken in account in strategies adopting elements of FBM. The selection of TLM and FBM strategies need to account for various factors that have otherwise been unaddressed in this study, such as the physical characteristics of the coastal aquifer, the climatic conditions, socio-economic aspects such as level of unpermitted water extraction and policy enforcement, the sensitivity of coastal ecosystems, and the degree of development in agriculture practices. Future applications of this lumped-parameter model to USB can include using the most recent boundaries and geometry of the aquifer from a concurrent study of USB (Knowling et al., in prep.) and the most recent modelled recharge rates (from Chapter 4). This simple model can be used in comparison with a more complex three-dimensional model of seawater intrusion in USB to test different management approaches, and therefore the performance of both simple and complex models can be compared and used as tools to support groundwater management decisions.

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6 Conclusions

This study focused on the groundwater recharge processes affecting the Uley South Basin (USB), South Australia. The USB is a highly heterogeneous coastal aquifer, subject to pumping, seasonal rainfall and evapotranspiration stresses. A review of historical well hydrographs in the basin showed a decline in groundwater levels that approach sea levels in places, raising concerns of seawater intrusion and aquifer depletion. This led to questions regarding the accuracy of previously published recharge estimates, and the performance of adopted groundwater abstraction management practices in the basin. The research was conducted as follows. Firstly, a conceptual model of recharge based on existing field data was developed, and recharge was estimated based on field-based methods. Secondly, the influence of model conceptualisation on one-dimensional (1D) unsaturated zone modelling of recharge was tested, as it applies to USB conditions. Thirdly, the field-based recharge estimates and the most likely conceptual models of recharge were used in GIS-based 1D modelling to extend point recharge predictions to the basin scale. Finally, modelled recharge rates were applied to a simple lumped-parameter water balance model to evaluate the aquifer responses to different management approaches. The key outcomes from each component of the investigation are detailed in the following.

6.1 Recharge processes and field-based estimates

Existent field data, including rainfall and other meteorological observations, groundwater quality, groundwater isotopes and groundwater levels, together with analyses of the USB's physical conditions were used to develop a conceptual model of the recharge processes in the basin. A plausible description of recharge processes arising from the available field data is given as: (i) chloride in the system is essentially from atmospheric deposition, which is influenced by proximity to the ocean; (ii) a significant proportion of rainfall occurring in dryer months is probably subjected to complete evaporation, with precipitated salts carried into the aquifer during wetter months; (iii) the primary function of sinkholes and other preferential flow features at the basin scale appears to be to redistribute water into the unsaturated zone rather than cause direct recharge to the aquifer; and (iv) soil water originating from (iii) is likely subject to transpiration more so than evaporation.

The water-table fluctuation (WTF) and chloride mass balance (CMB) methods applied in this study are modified forms of previous standard applications. Modification to the WTF method involved a simple correction for the contributions of pumping reductions (during wetter periods) to water-table rise, producing a basin-averaged and time-averaged (1967–2007) WTF recharge estimate that was lower by up to 30% than the “traditional” WTF method. The CMB approach was adapted to consider the location of recharge relative to the position of wells, based on chlorofluorocarbon (CFC) ages and groundwater flow distances and directions, resulting in a basin-averaged recharge that was 9% to 32% lower than if the chloride deposition was considered at the well location. Although the accuracy of the alternative recharge estimations cannot be validated, the importance of considering local site conditions in applying these common approaches to recharge estimation was demonstrated. The results indicate that an overestimate in WTF-based recharge may arise from neglecting pumping effects, and over- or under-estimates in CMB-based recharge may occur if chloride deposition factors are ignored.

Estimates of the long-term and spatially averaged recharge to USB were (i) 53 to 70 mm/year from the CMB approach, and (ii) 47 to 128 mm/year from the WTF approach. Ranges reflect uncertainty in input parameters. The application of the WTF method provides a large range of possible recharge values (i.e. larger than the CMB approach) because of the large uncertainty in the specific yield across the basin, and therefore is of limited relevance for long-term average recharge estimation in USB. Nevertheless, it is the only method capable of providing information about the time variability of regional-scale recharge in USB, considering the available field data. The range of recharge results presented here is smaller than previous estimates (i.e. 40 to 200 mm/year) for the basin. This was achieved partly by the modifications to the conceptual model for Uley South recharge beyond those of previous studies (Harrington et al., 2006; EPNRM, 2009), and through improvements to the methods for analysing the field data to account for local factors. The use of the classic head-matching regional analyses for estimating recharge (Theis, 1937) was not considered relevant for this study because: (i) the hydraulic conductivity is highly variable across the basin, and therefore this analysis would suffer from the inherent uncertainty associated with an inability to differentiate recharge variability from hydraulic conductivity variability; and (ii) this method would give an indication of pre-development, steady-state, spatially averaged recharge rates, while the intention with this study was to obtain transient, spatially-variable, recharge rates covering timeframes from pre-development to contemporary conditions.

6.2 Conceptualisation of one-dimensional modelling of recharge

1D simulations of unsaturated zone flow were used to explore USB recharge processes. These provide insights into the effects of various factors, such as vegetation, lithology, depth to water table (DWT) and preferential flow mechanisms, on spatial and temporal trends in recharge. The spatial variability of some of the factors controlling recharge is known or can be estimated with a reasonable degree of confidence (e.g. DWT or vegetation type). For others, such as rooting depth, sinkhole depth and lithology, the variability is largely unknown. To investigate the relative importance of both well characterised and poorly understood system characteristics, simulations using the LEACHM code were undertaken. In order to achieve this, LEACHM was modified to account for preferential flow in the unsaturated zone. For the hydraulic properties and water retention characteristics that were used (obtained from a combination of measurements in field samples and literature values), it was found that vegetation type and root zone depth exert the strongest control on the recharge magnitude and timing, which is consistent with other published studies (e.g. Keese et al., 2005; Smerdon et al., 2010; Scanlon et al., 2010). The model simulations showed that the role of lithological complexity is more important under vegetated areas when transpiration is a significant component of the soil water balance, compared to un-vegetated conditions. This is because the lithological composition determines the amount of water retained in the unsaturated zone, and modifies the region where capillary rise influences unsaturated zone water movements. That is, the lithology modifies mainly the amount of water available to meet transpiration demands, and thus has a significant impact on recharge rates. DWT is important in determining the monthly timing of the recharge, but annual recharge rates are largely invariant with respect to DWT.

This study applies unsaturated zone modelling to test the effect of different conceptualisations on recharge, including such factors as (i) complexity of the unsaturated zone lithology, and (ii) runoff infiltration and preferential flow. By comparing the timing of the modelled seasonal recharge peak to field data, it appears that partially penetrating bypass flow is a prerequisite to explain the observed seasonality of the recharge. However, it is not possible to distinguish which of the conceptual models of subsurface lithology (i.e. simple or complex profiles) are more likely to better represent the system, because they all result in recharge rates that are consistent with the observed timing of recharge. No single scenario of surface runoff infiltration and lithology complexity can be identified as a representation of the basin's physical setting, nor is such a scenario likely to exist given the spatial variability within the basin. Therefore, variability in parameters, attributable to both spatial heterogeneity and uncertainties in parameters values, needs to be considered when modelling recharge to USB at the basin scale. The testing of conceptual models allows for the development of 1D models that can predict recharge at the basin scale in USB, accounting for temporal and spatial distributions of recharge rates. The results demonstrate the benefits of using modelling as a means of testing different conceptual models of recharge. This is an approach that is sporadically used in groundwater flow modelling, but only rarely in recharge modelling.

6.3 Combining field-based estimates and one-dimensional modelling to obtain basin-scale recharge rates for Uley South

Field-based recharge estimates, 1D modelling and spatial distribution of DWT, vegetation types, substrate types and topographic slope were integrated in order to obtain fine temporal and spatial distributions of recharge at basin scale. The LEACHM code was used as a 1D simulator linked to a GIS framework, referred to as LEACHMG. LEACHMG's code was modified to account for preferential infiltration into the unsaturated zone. This combination proved to be an effective approach to calculate temporally and spatially variable recharge rates to USB, because the model was able to reproduce field-based timing, spatially distribution and time/space averages of recharge. This was a successful approach given that there was very limited unsaturated zone data to parameterize the 1D model, and critically comparing field-based estimations and modelling results allowed for a validated recharge model. Because 1D modelling of recharge can produce a very large range of possible values that are often difficult to be verified (e.g. Scanlon et al., 2002a), the use of field-based estimates to validate model predictions is advocated, which is not a common practice in recharge studies. This is a useful approach that can be applied to other aquifers where little data exist to construct a recharge model, and where field-based estimates are available.

The GIS-based simulations here considered result in spatially and temporally averaged recharge of 77 to 89 mm/year for USB, and of 61 to 74 mm/year if an area of sand dunes is not considered. Presently, the geometry of USB is being re-assessed and the sand dune area is being considered, at least partially, to be providing recharge to basins outside of the natural boundaries of USB (Knowling et al., in prep.). The modelled recharge rates compare well with the field-based CMB estimates of 53 to 70 mm/year. The comparison of the field-based WTF with simulated timing of recharge, both on annual and monthly bases, showed that the best matches were found for the simulations $S_{7.5}$ (simple lithological profile, runoff infiltration to a depth of 7.5 m) and $C_{7.5}$ (simple lithological profile, runoff infiltration to a depth of 7.5 m). These simulations produced annual recharge rates that are similar to the WTF trend. The best simulations of recharge to USB (i.e. $S_{7.5}$ and $C_{7.5}$) produced temporally averaged recharge rates of 89 and 84 mm/year respectively. Excluding the sand dune area, the recharge rates are 74 mm/year and 69 mm/year, respectively. The methodology presented here can aid in quantifying the uncertainty of recharge modelling predictions that support management decisions, and can help to reduce uncertainty by guiding data collection efforts (e.g. groundwater data for field-based estimates; unsaturated zone and vegetation data for the model parameterisation). For example, for the USB, it is imperative that a better regional assessment of the root distribution and vegetation types be undertaken, as well as a better understanding of sinkhole depths, as these factors proved to be the most important in controlling both timing of recharge and long-term averaged recharge.

The comparison of CMB and simulated recharge rates on a well-to-well basis showed a mismatch between the two approaches. This is likely because the recharge areas of the wells do not coincide with the location of the wells. Although there were attempts to define recharge areas for the wells, this proved to be challenging due to the aquifer being highly heterogeneous and given the different well screen depths and intervals. Generally, wells indicating higher CMB recharge rates are mostly located down-gradient of zones of higher modelled recharge rates. This is evidence that recharge occurring at one point in the aquifer is measured as a groundwater chloride concentration at a point down-gradient (as expected), and may be seen as an indication of consistency between the two approaches. Similarly, for the best simulations of recharge $S_{7.5}$ and $C_{7.5}$, although the comparison of field-based and modelled timing of recharge showed a similar pattern, some mismatches could be identified. This is not necessarily an inconsistency, but rather a result of both (i) lack and poor quality of available field data, and (ii) water-table movements being affected by other processes than recharge (e.g. lateral movements), which is not considered in the 1D model.

6.4 Recharge estimates applied to a coastal aquifer management model

The addition of Chapter 5 to this thesis provides a useful extension and application for the recharge estimates of the project, in that it shows a good example of application of the modelled recharge rates in a groundwater abstraction management model of the aquifer, demonstrating the importance of reliable modelled recharge rates on aquifer management. The USB is used as a case study to test various operational strategies for managing extraction from a coastal aquifer. A non-adaptive flux-based pumping management (FBM) regime has been adopted previously for the USB, and the groundwater levels have fallen steadily across a 30-year period, approaching sea level in some places. Furthermore, the current recharge rate of

140 mm/year adopted in setting groundwater management policies for the basin is likely an overestimate of recharge, as it was shown in this study. A parsimonious water-balance model is adopted to explore different pumping management approaches in the USB. The results indicate that the addition of a trigger-level management (TLM) approach to the FBM-based management of the system leads to (i) enhanced water availability manifested as higher allowable pumping volumes and fewer zero-pumping months; (ii) reduction in the risk of aquifer degradation (i.e. reduced sojourns/risk of seawater intrusion) and protection against recharge estimate inaccuracies; (iii) enhanced understanding of basin functioning leading to adaptive management. The present study also demonstrates the importance of performance-based (in terms of aquifer health) adaptation in the allocation of pumping; adaptive scenarios provided optimal water supply and resource security outcomes.

The USB case study, in combination with the review of previous studies, demonstrates that rigid flux-based approaches to coastal aquifer management may not adequately protect the aquifer from degradation nor provide optimal extraction volumes. The lack of TLM measures in reported accounts of aquifer management needs to be addressed, although trigger-level constraints need to be implemented judiciously, likely using a tiered approach, to avoid both over-regulating the aquifer and unnecessarily reducing available extraction. Careful consideration should be given to the application of water level and salinity trends as trigger levels (rather than just absolute values) in developing coastal aquifer management strategies. In addition, given that the risk of seawater intrusion is closely related to the discharge of the aquifer to the sea, this component of the water balance should be taken in account in strategies adopting elements of FBM. Changes in volumes of seawater in the aquifer were also shown to be an important element of the USB water balance. The selection of TLM and FBM strategies need to account for various factors that have otherwise been unaddressed in this study, such as the physical characteristics of the coastal aquifer, the climatic conditions, socio-economic aspects such as levels of unpermitted water extraction and policy enforcement, the sensitivity of coastal ecosystems, and the degree of development in agriculture practices.

6.5 Future work

An important outcome from this work is guidance into better characterisation of recharge to Uley South in future studies, but also, pertinent research questions to be addressed in the fields of groundwater recharge and coastal aquifer management are highlighted that have not been comprehensively explored in the scientific literature.

The uncertainty in field-based recharge estimates to USB can be reduced through further field data collection. Future efforts should aim at characterising the influence of vegetation in atmospheric chloride deposition to the basin, and therefore on the application of CMB to estimate recharge, which can potentially be significant.

A 3D groundwater flow model could be developed and enhance the the application of the methods described in this thesis. For example, for USB, a groundwater model will permit defining a recharge zone for each well, which can allow for a more precise comparison between the CMB and modelling results, and for the validation of the ability of the model to reproduce spatial variability of recharge. Knowledge of observation well capture zones will also improve the application of the CMB approach more generally, by more accurately assigning chloride deposition rates to groundwater samples. Similarly, simulated well hydrographs from a calibrated groundwater flow model will facilitate an improved comparison between the recharge timings from the WTF method and the 1D recharge model, given that the areas over which WTFs apply are presently difficult to characterise, and because many of the water-level records are discontinuous and/or short-term, and wells are unevenly distributed across the basin. Most importantly, it will support a critical evaluation of the differences in recharge signal arising from the WTF and 1D modelling, as they represent different physical processes.

Future applications of the lumped-parameter model of USB should include using the most recent boundaries and geometry of the aquifer (from a concurrent study of USB by Knowling et al., in prep.) and the most recent modelled recharge rates (from Chapter 4 of this thesis). The simple lumped-parameter groundwater model can be compared with a more complex 3D model of seawater intrusion of USB to test different groundwater management approaches. This will allow for improved understanding of the limitations of the simple approach as a tool to support groundwater management decisions.

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