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Impacts of Prolonged Drought on Salt Accumulation in the Root Zone Due to Recycled Water Irrigation

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Abstract Continuous use of recycled water (treated sewage effluent) over a long period of time may lead to the accumulation of salt in the root zone soil. This is due to the relatively higher levels of salt content in the recycled water compared to surface water. In this study, a laboratory column study was carried out to validate the HYDRUS 1D model under no rain condition. During the validation, the relative error and the % bias between observed and simulated soil water electrical conductivity (EC_{SW}) were found to be low and varied in a range of 5–10 and 5–6 %, respectively. The validated model was then used to predict long-term (5 years) salt accumulation under drought conditions. The analysis of model predicted salt values showed a cyclical pattern of salt accumulation in the root zone, and this related to the variation in rainfall and evapotranspiration. The mean root zone EC_{SW} in the 5th year was found to be within the highest salinity tolerance threshold for pasture (11.2 dS/m); however, the maximum root zone EC_{SW} was found to be about 63 % more than the threshold. Irrespective of seasons, in 5 years time, EC_{SW} at the depth of 1.0 m increased from 3.0 to 7.0 dS/m, which may pose a salinity risk to the groundwater table if there is a perched water table at a depth <1 m below the field surface. One of the management options to minimise long-term salt accumulation was also examined. By reducing the salt in recycled water by 50 %, it was possible to keep the EC_{sw} within the recommended threshold values. Overall, the methodology developed in this study can be used to identify appropriate management options for sustainable recycled water irrigation.

Keywords Soil water electrical conductivity · Hawkesbury Water Reuse Scheme · HYDRUS 1D · GS3 sensor

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1 Introduction

Recycled water use in irrigation schemes is receiving increasing attention by agriculturists. Significant benefits include a high nutrient content and the sustainability of reusing wastewater. However, there are also several concerns related to environmental and health risks. One such concern relates to the increase of salinity including sodicity and bicarbonate hazards in irrigated fields. Salinity is the concentration of soluble salts in water that are measured as total dissolved salts or electrical conductivity in soil solutions. From an environmental point of view, sodium and chloride are the two constituents of recycled water which are of most concern as they are more likely to remain as ions in soil solutions and contribute to the

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effects of salinity on plant growth (NRMMC-EPHC-AHMC 2006). The main contributors to salinity in sewage from the domestic sources are sodium-based detergents and other chemicals used in washing clothes and utensils and sodium-based salts used in food preparation (Patterson 2004; Stevens et al. 2011). In a typical household, washing machine contributes highest salt load followed by wastewater stream from toilets (Rahman et al. 2014a). In general, the salt along with other household waste (faeces, paper and food scraps) is discharged into sewer system and, finally, to the sewage treatment plant (Patterson 2004). While most organic matter is removed by various wastewater treatment processes, the majority of mineral salts pass through the wastewater treatment system unaffected, unless reverse osmosis is used as one of the treatment processes (Rebhun 2004).

As such, the recycled water contains elevated levels of salt; there is a potential risk of salt increase in the vadose zone (the unsaturated earth located between the ground surface and water table) when it is used for irrigation. As water evaporates from soils or is used by the plants, salts are left behind, a phenomenon which increases the concentration of salts in soil over time. The increase in salt concentration in the soil can adversely influence the amount of water a plant can uptake from the soil due to the osmotic effect. Several studies have reported increased salinity levels in soil due to the prolonged use of recycled water for irrigation (Dikinya and Areola 2010; Jahantigh 2008; Klay et al. 2010; Adrover et al. 2012; Marinho et al. 2013, 2014).

Abiotic stresses such as drought and salinity affect the plant growth and crop production (Wang et al. 2003). The probable reason of this growth reduction is water deficit or osmotic effects imposed by drought and salinity by reducing the soil water potential (Yuncai and Schmidhalter 2005). When the drought develops, the expected rainfall fails to occur at the expected time, which causes loss of soil moisture, surface runoff and groundwater recharge. Drought in Australia is a recurring phenomenon, with the most recent occurrence between 2000 and 2006 (also called 'millennium drought') (Nicholas et al. 2008). While different research focused on the impact of this drought period on groundwater–surface water interaction (Tweed et al. 2009), freshwater ecosystems (Nicholas et al. 2008) and fluctuations of water table in irrigation areas (Khan et al. 2008), there have been limited studies looking at the impact of prolonged droughts on salt accumulation in root zone due to recycled water irrigation.

In this research, impact of prolonged drought period on salt accumulation in a paddock was investigated. As a case study, a paddock (D21) was selected in the Hawkesbury Water Reuse Scheme (HWRS) situated in the southeastern part of Australia (Fig. 1). The Hawkesbury Water Reuse Scheme receives treated effluent from Sydney Water's Richmond sewage treatment plant and stores it in an on-site storage dam. The HWRS has always been under the attention of the scientific community of the western Sydney region because of its long-term recycled water use for irrigation, and consequently, several risk assessment studies related to recycled water irrigation have been carried out. This includes a health risk assessment (Derry et al. 2006), risk perception for using the recycled water for irrigation of a sports field and food production (Derry and Attwater 2006), development of a risk communication toolkit (Attwater et al. 2006), a study on the impact of recycled water treatment upgrade on the water quality in on-site storages (Aiken et al. 2010), investigation of regrowth of faecal indicator in the on-site storage dam (Derry and Attwater 2014) and the statistical analysis of water demand for irrigation (Stewart 2006). A limited study has been conducted focusing on the changes in soil properties due to recycled water irrigation and accumulation of salt in different sampling years (Aiken 2006). However, none of the previous studies have attempted to model the salt accumulation in the soil of paddocks of HWRS due to applications of recycled water for irrigation under drought condition. The main objective of this study is to apply a salt transport model for predicting the salt accumulation in the vadose zone soil when recycled water irrigation is practiced over a long period of time, say 5 years.

2 Materials and Methods

2.1 Soil Sample and Column Construction

Soil samples were collected from the D21 paddock (S 33° 37.478′ E 150° 45.706′), which has been irrigated under the HWRS. Soil samples were collected from the paddock between the depths 0 and 0.2 m using open pit method. The soil samples were transported to the lab, roots and worms were removed and samples were sieved using a 2.36-mm sieve. The soil was then air-dried at room temperature for 3 days. The air-dried

Fig. 1 Map showing location of D21 paddock within the campus of University of Western Sydney, Hawkesbury

samples were tested for different physical and chemical properties. The soil properties determined are summarised in Table 1. As can be seen in Table 1, the soil is of loamy sand and has saturated electrical conductivity (EC_e) of 0.375 dS/m. Soluble cations were determined according to Rayment and Higginson (1992) (method 14H1), where soluble Na^+, K^+, Ca^{2+} and Mg^{2+} were measured in saturated paste extract of D21 paddock soil sample, using atomic absorption spectrophotometer (AAS). The values of Na^+, Mg^{2+}, K^+ and Ca^{2+} were 1.17, 1.51, 0.56 and 0.29 meq/L, respectively. The sodium adsorption ratio was calculated as 1.2.

Table 1 Input parameters of HYDRUS 1D model for modelling salt accumulation in columns

Description	Value
Depth of soil in the column	0.47 m
Simulation period	103 days
Hydraulic model	VG-Mualem
Soil type	Loamy sand: sand=88.1 %, silt=6.0 %, clay=5.9 %
Bulk density	1511 kg/m^3
Water flow parameters	θ_r=0.041 m^3/m^3, θ_s=0.497 m^3/m^3, α=0.006, n=2.572, K_s=264.85 cm/day
Longitudinal dispersivity	1.3 cm^{-1} (Vanderborght and Vereecken 2007)
Initial condition	θ=0.09 m^{3}m^{-3}, EC_{SW}=2*EC_e (Ayers and Westcot 1985; Stevens et al. 2008), EC_e=0.375 dS/m
Boundary condition	Upper BC: atmospheric with surface layer Lower BC: free drainage
Type of transport model	Equilibrium model
Molecular diffusion coefficient in free water	1.75 cm^2/day (James and Rubin 1986)
Irrigation water EC	0.83 dS/m
Partitioning coefficient, k_d	0
Meteorological parameter	Recorded in laboratory

Three soil columns (C1, C2 and C3) of identical dimensions were prepared. Soil samples were packed into 600-mm-long columns constructed from 2.5-mm-thick plexiglass tubes. The columns had an external diameter of 160 mm. A 200×200 mm baseplate was used at the bottom of the column. The base plate was perforated with 2.0-mm diameter holes. A plastic mesh (mesh size <800 μm) was used at the bottom of the column. Figure 2 shows the schematic of the column setup. One of the three columns (C3) was fitted with two GS3 sensors at depths of 0.1 and 0.35 m from the soil surface. This was done to monitor bulk electrical conductivity at these two specified depths (Fig. 2). The whole setup was established inside a laboratory at Kingswood campus of the University of Western Sydney.

The GS3 sensors were connected to a data logger (CR800, manufactured by Campbell Scientific) for continuous data collection at minute intervals. Two soil water samplers were installed at the same depths where the GS3 sensors were installed and were used to collect soil water at the specified depths. The electrical conductivity of collected soil water was measured and recorded as EC_{SW}. All three columns were packed in such a way that an identical bulk density of 1511 kg/m^3 was maintained. Recycled water (RW) was collected from an on-site storage dam at Hawkesbury campus. The electrical conductivity of the RW was measured as 0.83 dS/m. The yearly mean values of cations in recycled water supplied to HWRS were collected from Sydney Water through personal communication. The values of Na^+, Mg^{2+}, K^+ and Ca^{2+} were 4.17, 1.72, 0.43 and 0.69 meq/L, respectively. The sodium adsorption ratio was calculated as 3.81. Meteorological parameters such as temperature, relative humidity and wind speed were monitored continuously by a weather station.

2.2 Calibration of the GS3 Sensor

The GS3 sensor is a dielectric sensor commonly used to quantify bulk electrical conductivity, volumetric water content and temperature. It is based on the principle of measurement of bulk electrical permittivity and related to the volumetric water content by a calibration equation (Topp et al. 1980; Vogeler et al. 1996; Munoz-Carpena et al. 2005). The sensor has three prongs which are 55 mm in length, 3.26 mm in diameter and 25.4 mm

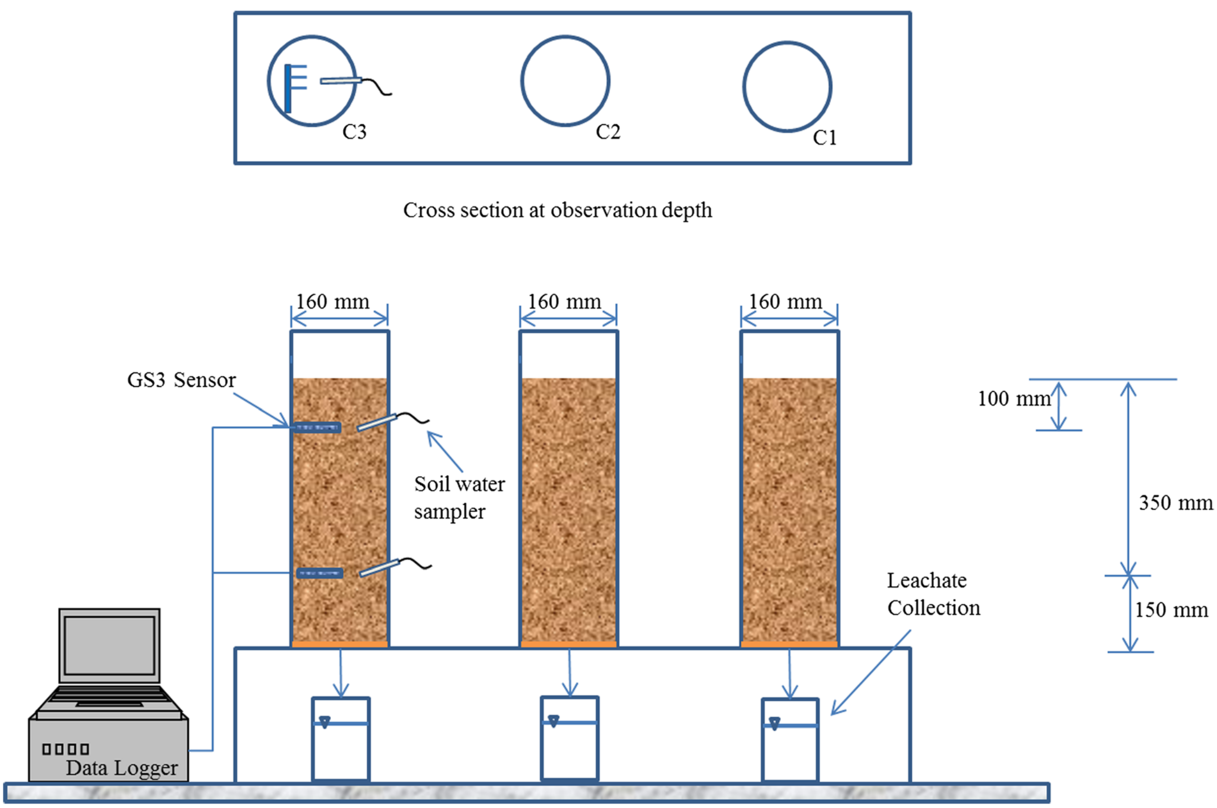

Fig. 2 Schematic of column setup

apart from each other. The detailed specifications of a GS3 sensor can be found in Decagon (2011).

A similar procedure to the one used by Munoz-Carpena et al. (2005) was used to calibrate the GS3 sensors. The air-dried soil was thoroughly mixed by hand with a known volume of synthetic water to achieve a uniform distribution of water and solute. The synthetic water was prepared by mixing four different salts, namely sodium chloride (0.01 mol), magnesium chloride (0.005 mol), calcium chloride (0.001 mol) and potassium chloride (0.001 mol) in 1 L of distilled water to produce a solution with an electrical conductivity of 2.0 dS/m. The ratio of cations (Na^+/Mg^{2+}/K^+/Ca^{2+}=0.6:0.2:0.1:0.1) in the synthetic water was maintained similar to that present in the recycled water. Three different types of synthetic water were prepared with electrical conductivities of 0.5, 1.0 and 2.0 dS/m. Each type of synthetic water was used to prepare duplicate soil samples with three different volumetric water contents, namely 0.2, 0.28 and 0.33 m^3/m^3. The soil was packed into a steel column of height 52 mm and diameter 98 mm. Mean packing bulk density was 1540 kg/m^3, which is similar to that measured at the field. Firstly, bulk electrical conductivity (EC_{bulk}) and permittivity (ε) were recorded by GS3 sensors for several minutes. Secondly, a subsample was collected by a volumetric soil sampler (height 38.8 mm and diameter 13.2 mm) to analyse the volumetric water content gravimetrically (method 2A1; Rayment and Higginson 1992). Thirdly, the electrical conductivity of the soil solution (EC_{SW}) was obtained by extracting the solution with a soil water sampler (Slim tube, manufactured by Soil Moisture Equipment Corp.) at suction 60–80 kPa and then reading the value with a laboratory EC meter (HACH Inc.). This procedure was repeated 18 times for all the soil samples which were prepared as explained above.

2.3 Column Experiment

The soil column experiment was conducted for a period of 103 days. Irrigation (recycled) water was applied at a frequency as per the current practice. Leached water at the bottom of each column was collected, measured for volume and analysed for electrical conductivity with an EC meter. Soil water samples were collected from the column at depths 0.1 and 0.35 m, as shown in Fig. 2, using the samplers at the end of each week. The volume and electrical conductivity of collected soil water samples were measured. Electrical conductivity of the soil water was temperature compensated according to USSL (1954).

2.3.1 Mass Balance of Salt

The cumulative mass of leached salt (g/m^2) was calculated by multiplying total dissolved solids (TDS) in leached water (g/m^3) by the amount of leached water (m). Salt concentration in the leached water was measured in terms of electrical conductivity (dS/m) and converted to TDS by using a multiplication factor of 640 (Stevens et al. 2008; Tchobanoglous and Burton 1991). The leaching fraction was determined by dividing the volume of leached water from the soil column by the total volume of applied irrigation water (USSL 1954).

2.3.2 Measures of Goodness of Fit

In addition to the visual comparison, four statistical parameters, namely mean absolute error (MAE), root mean square error (RMSE), % relative error (RE) and percent bias (PBIAS), were used to evaluate the goodness of fit between measured and predicted data.The mean absolute error (MAE) between the observed and predicted values is given by:

$$\mathrm{MAE} = \frac{1}{N}\sum_{i=1}^{N}|O_i - P_i| \tag{1}$$

where O_i represents observed values, P_i represents predicted values and N represents the number of observations. An MAE value close to 0 indicates better prediction by model. Similarly, the RMSE can be calculated by:

$$\mathrm{RMSE} = \sqrt{\frac{\sum_{i=1}^{N}(O_i - P_i)^2}{N-1}} \tag{2}$$

MAE and RMSE indicate the presence and extent of deviation between the simulated and observed values (Ramos et al. 2011; Kanzari et al. 2012). Units of MAE and RMSE are the units of that particular variable.

RE measures the relative error in the simulated values in terms of percentage with respect to observed values. An ideal value of RE is 0, which indicates the simulation result is perfect and can be calculated by:

$$\mathrm{RE} = \frac{100}{N}\sum_{i=1}^{N}|O_i - P_i|/O_i \tag{3}$$

PBIAS measures the percentage of the residuals with respect to observed values which indicate whether the model overestimates or underestimates the observed values (Moraisi et al. 2007). The perfect value of PBIAS is 0. Low values of PBIAS indicate better simulation results by the model whereas positive and negative values represent underestimation and overestimation of bias, respectively, in the simulated results. PBIAS can be calculated by:

$$\mathrm{PBIAS} = \frac{\sum_{i=1}^{N}(O_i - P_i)}{\sum_{i=1}^{N}(O_i)} \times 100 \tag{4}$$

2.4 HYDRUS 1D Model

The HYDRUS 1D model was used to simulate one-dimensional water flow and solute transport in incompressible, porous, variably saturated soil under transient system. In HYDRUS 1D, water flow modelling was described by Richards' equation. Similar to other studies (for example, Bunsri et al. 2008), for Hydraulic properties, Van Genuchten's equation (1980) was used, which provides relationships between the volumetric moisture content, effective saturation, hydraulic conductivity and specific moisture storage. For solute transport, it was assumed that the solutes were non-reactive and there was no solubilisation or dissolution of soil minerals. This assumption enabled modelling of the salinity in the soil based on the convection–dispersion equation for non-reactive solutions (Roberts et al. 2009). The governing water flow and solute transport equations were solved using the upstream weighting finite element method. A full description of the model is given by Simunek et al. (2009).

2.4.1 Input Parameters of HYDRUS 1D

The input parameters required by the HYDRUS 1D model were collected from different sources. The model requires input parameters such as physical, hydrological and solute transport characteristics of soil profile, daily standard meteorological data as well as crop information. The input parameters are summarised in Table 1.

Physical characteristics of the soil such as textural analysis (NCST 2009) and saturated electrical conductivity (EC_e) were determined in the laboratory. Bulk density was measured at the time of packing the columns. Water flow parameters were evaluated by fitting laboratory-measured soil water characteristics data using the RETC software package (Van Genuchten et al. 1991). The soil water characteristics curve was evaluated by pressure plate method (ASTM 2002). Solute transport parameters were collated from literature and are given in Table 1.

One of the important input parameters for HYDRUS 1D for transient modelling of salt transport is time variable boundary conditions, including precipitation, evaporation and transpiration. For the column study, time variable boundary conditions were determined from meteorological parameters measured in the laboratory. Daily values of potential evaporation (ET_0) were calculated using Penman-Monteith method by HYDRUS 1D (Simunek et al. 2009). Daily values of ET_0 were in the range of 0.7–1.0 mm/day (Fig. 3). Generally, there was not much variations in the minimum (18 to 23 °C) and maximum (19.6 to 24.9 °C) temperatures and solar radiation in the laboratory. Solar radiation was measured in watt per square meter. The values of 1-day measurement were averaged over a day and reported as millijoules per square meter per day. The solar radiation varied between 2.96 and 3.17 MJ/m^2/day. A significant variation in the relative humidity was observed, which ranged from 30 to 60 %. The wind speed in the laboratory depends on the flow from the air-conditioner, working between 6 a.m. and 6 p.m. over the five working days (Monday–Friday). During the weekend, the air-conditioner was not running. Rainfall was not considered in the column study. Irrigation scheduling included applying recycled water three times per week (Fig. 3). Initially (up to day 9), a relatively higher amount of recycled water was applied. These higher applications were required to soak the soil in the column. In the first 9 days, 270 mm of recycled water was applied, after which the columns were kept dry for 13 days. On average, 97.2 mm of irrigation water was applied per month.

3 Results and Discussion

3.1 Soil-Specific GS3 Sensor Calibration

Results of the calibration of volumetric water content are shown in Fig. 4, where GS3 sensor-measured permittivity (ε) is plotted against gravimetrically determined volumetric water content (θ). An empirical calibration curve was obtained, to determine θ from permittivity, by fitting a quadratic function to the data, which is:

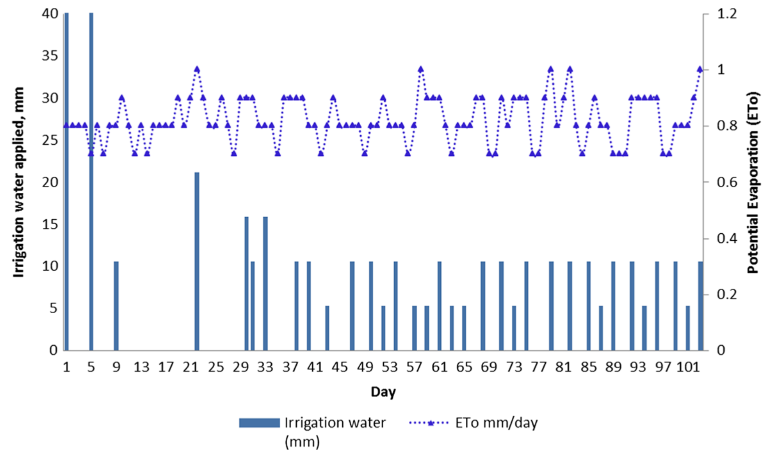

Fig. 3 Variation of ET_0 and irrigation water applied (days 1 and 5 have irrigation amounts of 212 and 53 mm, respectively)

$$\theta = -2.0 \times 10^{-5}\varepsilon^3 + 5.0 \times 10^{-4}\varepsilon^2 + 1.23 \times 10^{-2}\varepsilon + 5.26 \times 10^{-2} \quad (5)$$

This evaluated empirical relationship deviates from the relationship suggested by Topp et al. (1980) by a maximum value of 0.1 m^3/m^3 (RE=8.1 %). The volumetric water content (θ) obtained using the equation of Topp et al. (1980) is also shown in Fig. 4 (broken line). As shown in the figure, the θ values obtained using the equation of Topp et al. (1980) deviate significantly from the observed values for higher permittivity values. A similar observation was also made by Vogeler et al. (1996), which the authors attributed to the difference in soil texture including bulk density, clay content and organic matter present in the soil. As can be seen in Fig. 4, Eq. 5 appears to explain observed θ values satisfactorily. As such, it is suggested that Eq. 5 may be used for calculating θ, instead of the one proposed by Topp et al. (1980) for HWRS soil.

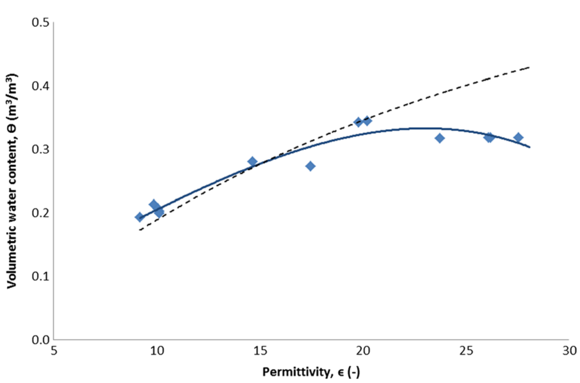

Fig. 4 Relationship between gravimetrically determined water content and GS3 recorded Permittivity. The *solid line* shows the fitted quadratic function and the *broken line* shows the empirical relationship suggested by Topp et al. (1980)

Results of the soil water calibration are shown in Fig. 5, where the electrical conductivity of soil water is plotted against the GS3 sensor-measured bulk electrical conductivity for different volumetric water content.

From the results of the calibration study, a regressed equation was developed for predicting EC_{SW} from sensor measured EC_{bulk} and θ data. In the literature, there are reported relationships between EC_{SW}, EC_{bulk} and θ (Vogeler et al. 1996; Nadler 1997; Munoz-Carpena et al. 2005); however, a soil-specific relationship between these parameters is required for better estimation of EC_{SW}. The regressed equation is:

$$EC_{SW} = -1.10 + 1.62 \times \ln(EC_{bulk}) - 4.25 \times \ln(\theta) \quad (6)$$

The coefficient of correlation (R^2) for Eq. 6 was found to be 0.94. The p value of both predictor variables, i.e. EC_{bulk} and θ of Eq. 6 was very low (p<0.0001). High R^2 and low p values suggest that Eq. 6 can be used to estimate EC_{SW}, the given EC_{bulk} and θ values. The calculated MAE, RMSE and RE for this equation were found as 0.14, 0.17 and 5.2 %, respectively. The developed Eqs. 5 and 6, from the calibration study, were used to calculate θ and soil water electrical conductivity (EC_{SW}) from data received from continuous real-time monitoring with GS3 sensors at depths 0.1 and 0.35 m.

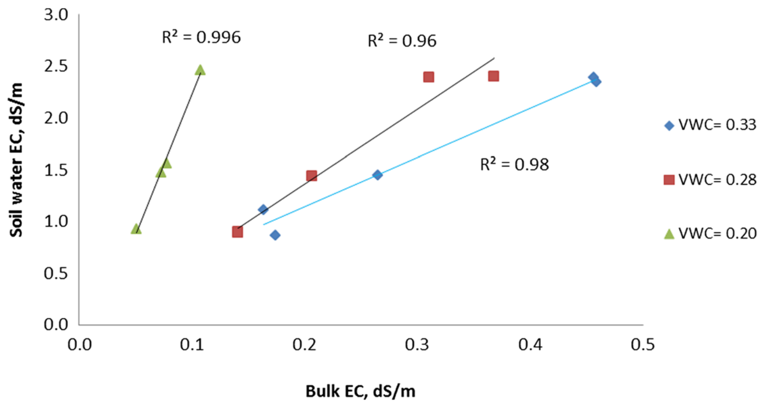

Fig. 5 Relationship between soil water electrical conductivity and the GS3 sensor-measured bulk electrical conductivity for different volumetric water content. Fitted relationships are shown in *solid line*

3.2 Salt Mass Balance as an Indicator of Salt Accumulation

Mass balance calculation of solute in a soil profile provides general information regarding accumulated salt in the profile. The results presented in this section aid an understanding of the general trend of salt accumulation in the soil profile without considering the specific depth.

Figure 6 shows applied, stored and leached salt loads during the study period. Leached salt load from all three columns varied with a standard deviation of 0.4 to 2.7 g/m^2, except for the first day, which was 6.9 g/m^2 (results not shown). This variation in the leaching of salt can be attributed to variations in the packing of the columns. Throughout the study period, the total cumulative leached salt mass (averaged for three columns) was less than the total cumulative applied salt mass (Fig. 6). Thus, the total cumulative salt mass stored in the soil profile showed an increasing pattern.

Leaching of salt is considered one of the irrigation management options by field managers (Corwin et al. 2007). Conventionally, in the field, leaching fraction (LF) is used to calculate the salt buildup in the soil (Ramos et al. 2011). Results from the column study shows a strong correlation (R^2=0.95) between salt accumulation in the soil profile and leaching fraction was observed, which is shown in Fig. 7. Results from Fig. 7 show that salt buildup in the soil profile decreased with increasing LF. A similar observation was reported by Corwin et al. (2007) and Duan et al. (2011). However, no correlation equation between these two variables (i.e. salt buildup and LF) was proposed because LF is not the only parameter associated with salt buildup in a soil profile. Other factors such as evapotranspiration and rainfall are important and should be considered when calculating salt accumulation in a soil profile.

3.3 Validation of HYDRUS 1D Model with Column Study Result

In the previous section, the salt mass balance provided a picture of salt accumulation in the soil profile. The

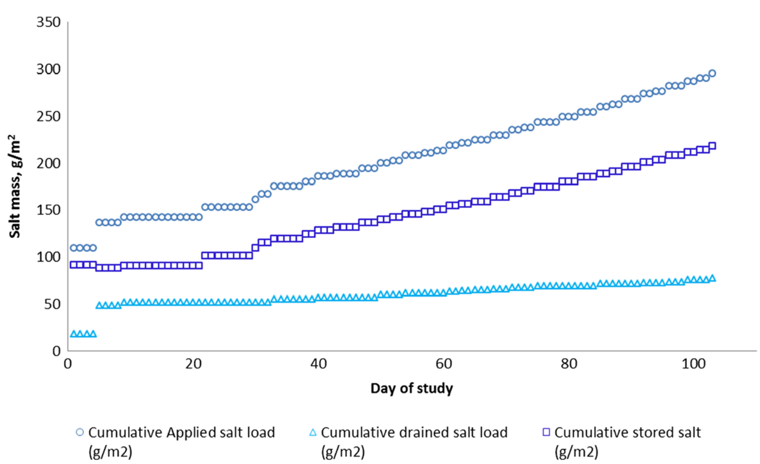

Fig. 6 Cumulative salt mass applied, salt mass leached and salt mass stored in the column profile (averaged over the results from three columns)

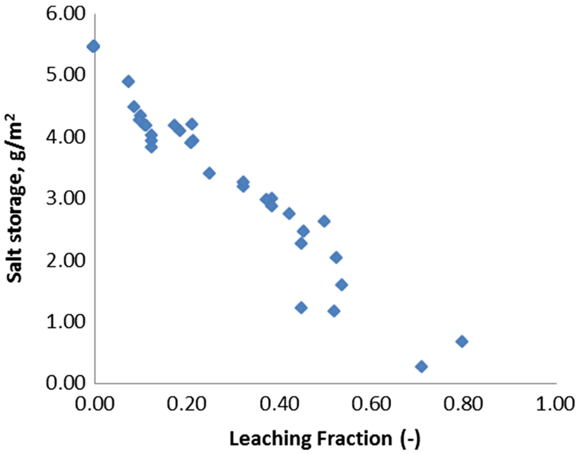

Fig. 7 Controlling salt buildup in soil profile by leaching fraction

absolute salinity level and the spatial distribution of salinity throughout the soil sample were, however, unable to be evaluated. In situ collected soil water samples and data recorded from the sensors were useful in determining the spatial distribution of salt accumulation, which is presented in this section. In addition, the observed data was used to validate the HYDRUS 1D model.

HYDRUS 1D is validated by different studies simulating different scenarios (Kanzari et al. 2012; Ramos et al. 2011; Sutanto et al. 2012; Sarmah et al. 2005; Phillips 2006) with a satisfactory outcome. In some studies, HYDRUS 1D was calibrated by inverse modelling (Kanzari et al. 2012; Sutanto et al. 2012) with observed data from part of the study period. Through inverse modelling, soil hydraulic modelling parameters were obtained which were used for validating the model in combination with the observed data of the rest of the study period. However, in this study, soil hydraulic or water flow parameters were determined in the laboratory (Table 1) and used in the HYDRUS 1D model. Therefore, after a successful validation study (discussed under this section), the model would be assumed as calibrated, validated and able to be used with confidence to predict salt accumulation.

The in situ measured EC_{SW} and the one predicted by HYDRUS 1D are shown in Fig. 8a for the depths 0.1 and 0.35 m on all sampling occasions. At both the depths, an increasing pattern of EC_{SW} over time was observed (Fig. 8b). The EC_{SW} measured in situ was relatively higher at the depth of 0.1 m compared to the value at depth 0.35 m. The EC_{SW} increased by about 1.9 times above its initial value (from 1.0 to 1.9 dS/m) at the depth of 0.35 m and about two times (from 1.0 to 2.0 dS/m) at the depth of 0.1 m. EC_{SW} increased substantially at 0.1 m depth between 5 and 60 days, then stabilised. EC_{SW} at 0.35 m depth was stable until about 70 days, and then increased gradually to 100 days. This shows that salt accumulation shifts from shallower to deeper depths as the irrigation continues. Initially, EC_{SW} is higher at the shallower depths due to lower volumetric water content owing to evaporation (Rahman et al. 2014b). However, as the irrigation continues, the salt is transported to lower levels, thereby gradually increasing the EC_{SW} at deeper depths. The calculated MAE, RMSE, RE and PBIAS between in situ measured and simulated EC_{SW} are shown in Table 2. The MAE and RMSE decreased with depth, indicating better prediction of EC_{SW} by HYDRUS 1D at depth 0.35 m compared to predictions at depth 0.1 m. However, RMSE values for prediction of EC_{SW} for both the depths agreed with the range reported by other researchers (0.21 to 3.73 dS/m) who used HYDRUS 1D for salt transport modelling (Ramos et al. 2011; Kanzari et al. 2012; Yurtseven et al. 2013; Forkutsa et al. 2009).

The sensor-measured EC_{SW} (calculated by Eq. 6) for both the depths are also shown in Fig. 8a. As shown in Table 2, the correlation between in situ measured EC_{SW} and simulated EC_{SW} appears to be superior compared to the correlation between sensors measured EC_{SW} and simulated EC_{SW}. As such, in all the forthcoming analysis, in situ measured EC_{SW} values were used.

Agreement between simulated and in situ measured results on EC_{SW} strongly suggests that the HYDRUS 1D model can be used with confidence in predicting salt accumulation in paddocks for which it is validated. However, the small discrepancy between the observed and predicted EC_{SW} (in terms of different goodness of fit indices) might be because of the edge effect, preferential flow and locally entrapped air (Peck 1969), which is not considered by HYDRUS 1D. Effort was made to minimise these phenomena by using loamy sand soil and relatively wide columns, not fully drying columns during experimental cycles, and by positioning sensors and samplers at the central part of the column. However, the existence of cracks, roots and, sometimes, gaps between soil and column material may cause preferential flow, which is the reason for uneven and rapid solute movement in the soil (Phillips 2006). Edge effect and preferential flow may cause applied irrigation water to bypass soil matrix without accomplishing adsorption of salt causing its accumulation in the soil. Nevertheless,

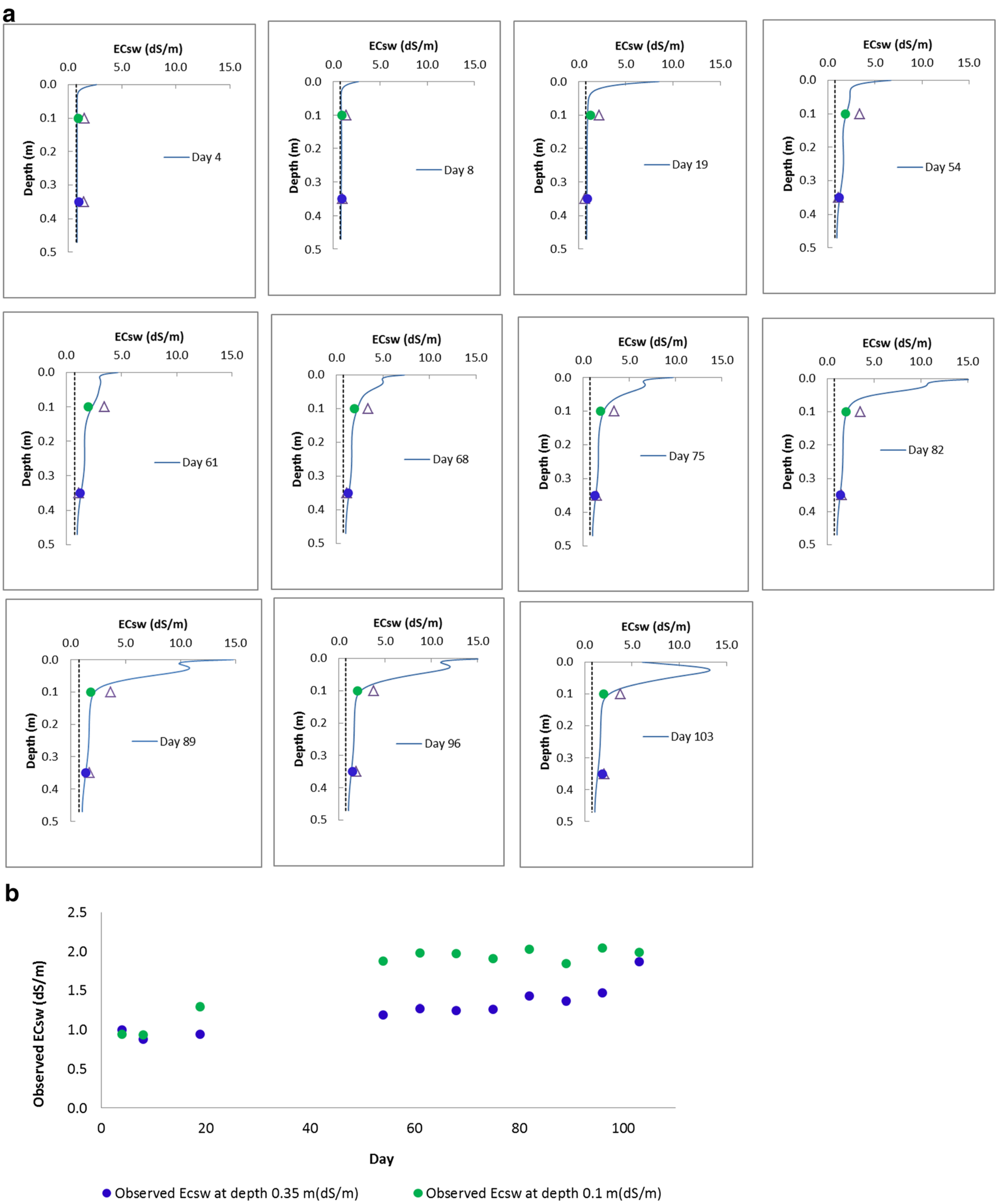

Fig. 8 **a** Measured and simulated soil water concentration (EC_{SW}) profile during different sampling time. *Dashed line* represents the initial value, *solid line* represents HYDRUS 1D prediction, *circle* represents in situ extracted EC_{SW} and *triangle* represents calculated EC_{SW} by Eq. 6. **b** Variation of in situ measured EC_{SW} with time at depths 0.1 and 0.35 m

Table 2 Results of the goodness of fit indices between observed and predicted EC_{SW} at different depths

Goodness of fit indices	0.1 m		0.35 m	
	In situ measured EC_{SW} vs. simulated EC_{SW}	Sensor measured EC_{SW} vs. simulated EC_{SW}	In situ measured EC_{SW} vs. simulated EC_{SW}	Sensor measured EC_{SW} vs. simulated EC_{SW}
RMSE (dS/m)	0.25	1.22	0.18	0.32
MAE (dS/m)	0.40	1.15	0.12	0.24
RE (%)	10.2	33.8	5.4	17.4
PBIAS	−6.5	36.1	5.0	9.1

the existence of preferential flow in column studies is reported by different researchers (Camobreco et al. 1996; Duan et al. 2011).

In this study, the bare soil column (without any vegetation) is used. This is because the focus of the study is to identify distribution of salt accumulation rather than effect on plant growth and crop yield. In the validation of the model, root water uptake was not considered; hence, potential evapotranspiration equals the potential evaporation. Therefore, bare soil column experiment was sufficient to get enough information to validate and calibrate the model with laboratory-measured soil hydraulic and water flow parameters.

3.4 Application of HYDRUS 1D for Modelling Salinity Levels in HWRS Paddock

3.4.1 Model Parameters

The validated HYDRUS 1D model, by column study, was used to predict long-term salt accumulation in the D21 paddock. For this purpose, the soil type, soil hydraulic model and parameters, boundary conditions for water and solute transport, type of transport modelling, molecular diffusion coefficient, partitioning coefficient, initial conditions for water content and soil water concentration and irrigation water salinity were kept identical to those used in the laboratory column study modelling. For the field prediction, a soil profile up to 1 m below the ground level is considered. Bulk density and longitudinal dispersivity was 1500 kg/m^3 (measured in the field) and 20 cm^{-1} (Vanderborght and Vereecken 2007), respectively. An irrigation schedule was calculated based on Allan et al. (1997) for ryegrass pasture in loamy sand soil. The maximum amount of irrigation water to be applied for irrigation was calculated based on the product of the average water holding capacity of loamy sand and root depth of ryegrass pasture. For loamy sand, water holding capacity was 55 mm/m (SARDI 2014) and the average root depth of ryegrass pasture was assumed as 0.35 m (Allan et al. 1997). The irrigation interval in a month was then calculated dividing the maximum irrigation by actual average monthly evapotranspiration. For this, a crop factor of pasture was used for different months varying between 0.4 and 0.7 (Allan et al. 1997). Using the methodAllan et al. (1997), a total of 47 irrigation events per year were calculated and distributed from January to December as 7, 5, 5, 3, 2, 1, 1, 2, 3, 5, 6 and 7, respectively, for each month. In each irrigation event, 19.25 mm of irrigation water was used (Fig. 9).

The long-term prediction over 5 years was carried out. As stated earlier, the purpose of this simulation was to observe the accumulation of salt during the drought period. It is expected that the drought period yields the worst case scenario, as under normal rainfall conditions the salt may be flushed from the root zone, thereby reducing the potential for salt accumulation under normal rainfall conditions (Rahman et al. 2014a). Meteorological data was collected from the weather station (station number 067021) at Hawkesbury campus, University of Western Sydney. To identify the minimum rainfall year of the drought period (2000 to 2006) within the preceding decade, rainfall data from 2001 to 2013 was statistically analysed and found out that the year 2006 had the least amount of rainfall compared to other years. The total amount of rainfall in the year 2006 was 525.20 mm. This amount of rainfall was about 15 % less than in the year 2005 and 49 % less than in the year 2007. The rainfall of other years of this decade varied within the abovementioned range. Mean annual rainfall for this

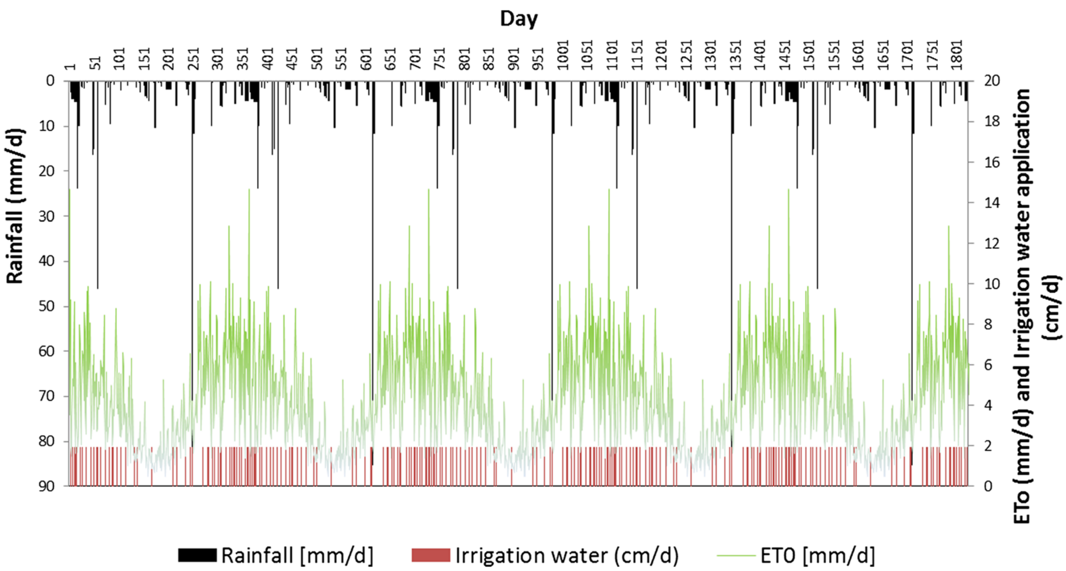

Fig. 9 Variation of ET_0 (mm/d), rainfall (mm/day) and irrigation water applied (cm/day) under drought condition

weather station for the period of 1881 to 2013 is 801 mm (BOM 2014), which is 53 % higher than the rainfall in 2006. Therefore, for the purpose of calculating maximum amount of salt accumulation (due to minimum rainfall), it was assumed that the climatic condition of the year 2006 would continue for a period of 5 years under drought condition. The climate condition of the year 2006 was used to calculate potential evapotranspiration (ET_p), which is shown in Fig. 9. In HYDRUS 1D, daily potential evaporation (E_p) and transpiration (T_p) are required as input data. The model then converts them into actual E_p and T_p based on the available soil moisture content. Potential evapotranspiration was calculated by the Penman-Monteith method (Simunek et al. 2009) with the data collected from the weather station. ET_p was then divided into potential transpiration (T_p) and evaporation (E_p) using Beer's Law (Wang et al. 2009):

$$E_p = ET_p \times e^{-k \times LAI} \tag{7}$$

$$T_p = ET_p - E_p \tag{8}$$

$$\mathrm{LAI} = 5.5 + 1.5\ln(h_c) \tag{9}$$

k is an extinction coefficient set to be 0.463[−] and LAI is leaf area index [LL^{-1}]. LAI was calculated from a crop height (h_c) of 0.3 m (Allen et al. 1998). The potential transpiration and evaporation varied in the range of 0.4 to 12 and 0.1 to 2.7 mm/day, respectively. Crop type (pasture), root water uptake model and root water uptake parameters were taken from the HYDRUS 1D built-in library. Plant solute uptake was assumed to be negligible in the present study.

3.4.2 Long-Term Prediction and Salinisation Risk

As explained earlier, the HYDRUS 1D model was applied for studying the possible impacts of recycled water irrigation on salt movement and accumulation in the paddock over the period of 5 years of continuous irrigation during drought condition. The results obtained from the simulation of 5 years of irrigation are presented in Fig. 10. The reported salt accumulation profile (Fig. 10a) represents the salt accumulation averaged over a particular year. For example, the year 1 profile shows an average variation of salt accumulation throughout the profile depth considering all days in the year. As expected, the soil salinity profile shows a cyclical pattern (alternatively increasing and decreasing with time) in the same year (Fig. 10b). The cyclical pattern of the salt accumulation in the soil profile is linked to the variation of rainfall and evapotranspiration (Devitt et al. 2007). Similar patterns of salt accumulation were also reported by other researchers (Kanzari et al. 2012; Kato et al. 2008; Thayalakumaran et al. 2007).

An increasing pattern of soil water concentration in the root zone (0 to 0.4 m) with years of irrigation was observed (Fig. 10a). With the increase of time, the rate of increase of root zone EC_{SW} decreased. From year 1 to

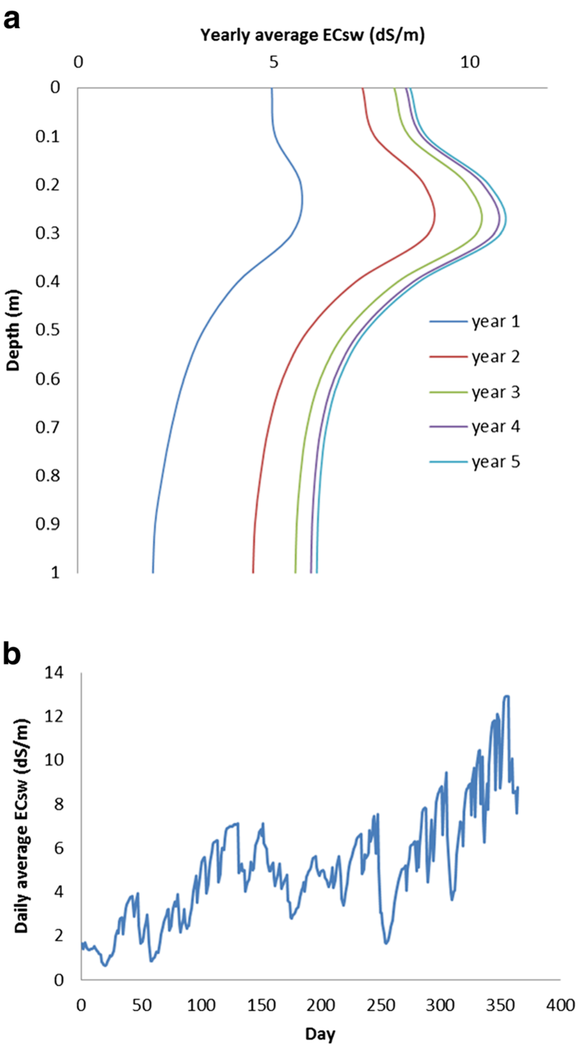

Fig. 10 **a** Long-term simulated soil water salinity profile for irrigation with recycled water under drought condition. **b** Cyclical pattern of daily average root zone salt accumulation in the year 1

the year 2, root zone EC_{SW} (averaged over 0 to 0.4 m) increased by 57 %. Similarly, between two subsequent years, i.e. years 2 to 3, 3 to 4 and 4 to 5, EC_{SW} increased by 13, 4 and 1 %, respectively. This trend of decrease of salt accumulation rate indicates that the system is trying to reach equilibrium by increasing the vertical transport (Allen et al. 1998).

Another aspect of the long-term simulation is to identify the risk of accumulation of salt at deeper levels of the soil profile, which is transported vertically. This is important because washed out salt from the root zone may end up contaminating groundwater aquifers. It is interesting to see from Fig. 10a that the use of recycled water for irrigation resulted in the increase of EC_{SW} with time at the depth of 1.0 m. From the year 1 to the year 5 of continuous irrigation, the yearly average EC_{SW} at the depth of 1.0 m was increased almost three times (from 1.9 to 6.1 dS/m). On average, in the lower portion of the soil profile (0.5 to 1.0 m), salt accumulation was less than that of root zone salinity. From year 1 to year 5, EC_{SW} in the lower portion of the soil profile was 53, 38, 34, 32 and 32 % less than that of root zone EC_{SW}, respectively. The results indicate more evapotranspiration in the root zone occurred than the lower portion of the soil profile and, over the period, migration of salt downwards.

3.4.3 Seasonal Variation of Salinisation in Root Zone

The D21 paddock is situated in the temperate climatic zone of Australia, where the seasons are divided into four groups, namely summer (December to February), autumn (March to May), winter (June to August) and spring (September to November) (Wells 2013). The seasonal variation of salinisation is important in the sense that this provides more succinct picture of salinisation than the yearly average of root zone EC_{SW} for a certain crop. The seasonal variation of root zone EC_{SW} is highlighted in Fig. 11. The seasonal variation of root zone EC_{SW} follows similar pattern of yearly salt accumulation, where salt accumulation is trying to reach equilibrium condition over 5 years period. Spring and summer seasons showed more salt accumulation than winter and autumn seasons throughout the simulation period. Initially, up to the second year, spring season showed more salt accumulation than summer season; however, at the fifth year, the summer season showed 2 % more salt accumulation compared to spring season and about 9 % more salt accumulation than autumn and winter seasons. This can be explained based on the application of recycled water for irrigation and

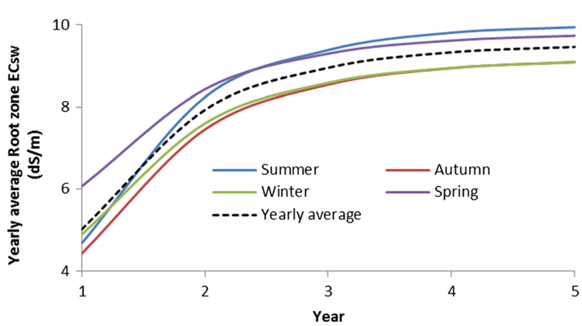

Fig. 11 Seasonal variation of simulated root zone salinity under drought condition

occurrence of rainfall. During spring and summer, as explained earlier, the field receives higher quantity of recycled water. This, coupled with relatively higher evapotranspiration during these two seasons, explains the higher levels of EC_{SW} in the root zone. The gradual change in the higher EC_{SW} to summer can be attributed to the rainfall, which incidentally is higher both in spring and summer. This is a result of progressive accumulation and leaching of salt within the root zone to reach a state of equilibrium. The yearly average seasonal variations of root zone EC_{SW} were found to be lower than salinity tolerance thresholds for pastures (in terms of EC_{SW}) including clovers and ryegrass. According to NRMMC-EPHC-AHMC (2006), the threshold varies from 3.0 to 11.2 dS/m. However, some monthly average root zone EC_{SW} was found higher than the maximum threshold of salinity tolerance especially in the month of December of the simulation years. In the first year, root zone EC_{SW} in December was 14 % higher than the maximum salinity threshold, which increased to 63 % (EC_{SW}=18.3 dS/m) in the fifth year.

3.4.4 Management Scenarios to Control Salinisation

It is recognised from the above discussion that long-term irrigation with recycled water progressively increased the salt accumulation in the root zone. Increased levels of salinity in the root zone may affect plant response in terms of leaf xylem water potential, tissue moisture content and colour of grass in open space (Lockett et al. 2008). Rengasamy (2006) and Grewal and Maheshwari (2013) observed that salinity in root zone reduces the root water uptake by plants, which reduces plant growth and, hence, reduces crop yield. This necessitates an examination of possible management options which can relieve the salt accumulation due to long-term irrigation using recycled water.

The management option considered in this research is to reduce the salt level in recycled water before using it in the irrigation. Simulations were carried out to find required amount of salt reduction in recycled water so that the salt accumulation in the root zone remains sustainable (i.e. below the maximum salinity threshold limit) while irrigating with recycled water. The simulation results are shown in Fig. 12. The figure shows a clear reduction of root zone salt accumulation due to 50 % reduction in the salt concentration (in terms of electrical conductivity) in recycled water. The root zone EC_{SW} at the fifth year reduced by 48.9 %. Similar percentage of EC_{SW} reduction occurred in the lower part (0.5 to 1.0 m) of the soil profile.

One of the main purposes of investigating this management option was to see if reducing the salt content in recycled water was able to reduce root zone EC_{SW} for all the months of a year (especially in the month of December). Impact of the recycled water treatment strategy (i.e. salt reduction) on the maximum root zone salt accumulation is shown in Table 3. It is evident from the table that 50 % reduction of salt in recycled water was sufficient to keep the maximum root zone EC_{SW} within the acceptable NRMMC-EPHC-AHMC (2006) threshold limit in all the years (years 1 to 5). It should be noted that reduction of the salt concentration in RW by 50 % reduced the EC_{SW} in soil water by about the same amount. It appears that the salt concentration in soil water solution vary linearly with that of recycled water, which is also reported by Rahman et al. (2014a).

Given that the conventional treatment system is unable to remove salinity from recycled water (Rebhun 2004), it may be necessary to install tertiary treatment system comprising reverse osmosis (RO) process to remove salt from the recycled water. Also, it is not necessary to treat all the recycled water with reverse osmosis process. For example, by blending or mixing recycled treated water using reverse osmosis with the equal amounts of recycled water that is not treated using recycled water can yield 50 % salt reduction in the irrigation water.

Thus, the proposed management option of partially treating recycled water using reverse osmosis process may be an appropriate solution to reduce salinity impacts on the ground water table, especially for the perched aquifer situated under this paddock in HWRS. Beveridge (2006) monitored the ground water table at this site from January 2004 to April 2005 and reported that the depth of the water table of the perched aquifer was between 1.4 and 2.4 m. Therefore, continuous irrigation using 100 % recycled water over a long period of time may impact on the salinity levels of the groundwater in the perched aquifer.

Instead of using RO-treated recycled water to blend with untreated recycled water, it is possible to blend harvested stormwater with untreated recycled water. Stormwater harvesting involves collecting, storing and treating stormwater from urban areas, which can then be reused (Sydney Water 2013a). Stormwater is expected to contain relatively less salt levels (electrical conductivity of 0.17 to 0.34 dS/m) as compared to the recycled

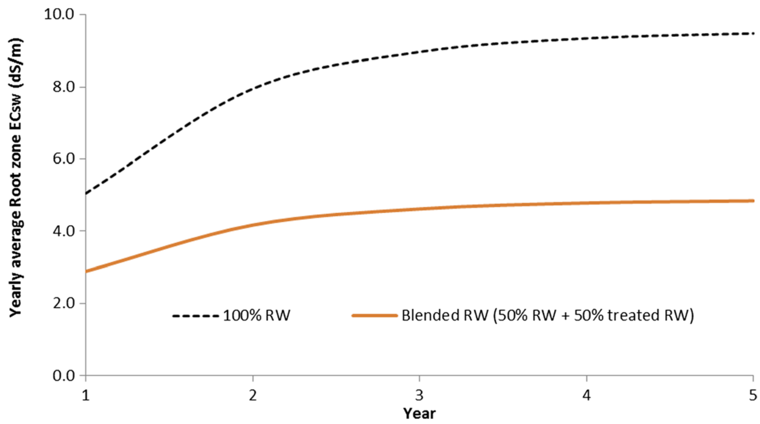

Fig. 12 Impact of salt reduction strategy in the recycled water on the root zone salt accumulation at different years of continuous irrigation

water (Sharpin 1995). A well-designed and managed stormwater management scheme, such as wetland and urban lake, may be able to supply required quantity of water for blending with recycled water. However, care should be taken to maintain the quality of the stormwater to be used. Especially during the prolonged drought period, the quality of detained stormwater should be monitored for possible increase of salinity. Moreover, during drought conditions, sufficient stormwater may not be available for use in the blending. Furthermore, economic feasibility must be considered before deciding to establish the stormwater harvesting scheme (Knights and McAuley 2009).

It is understood that conventional management options such as leaching fraction might be used to reduce the salt buildup, as explained in Fig. 7, because leaching fraction is highly correlated with salt buildup in the soil profile. Leaching fraction-based salinity management was helpful to reduce a high salinity level in open space in the lower Colorado River basin (Devitt et al. 2007). However, the leaching fraction was not considered as the sustainable option in this study because increased leaching flushes considerable amount of salt out of the root zone and into the groundwater aquifer (Khan et al. 2007); in the case of using blended recycled water, less amount of salt will be leached. Therefore, mixing treated (using RO) and untreated recycled water before applying as irrigation water is considered to be more sustainable. This solution was also recommended by Grewal and Maheshwari (2013) for continuous long-term irrigation using recycled water.

4 Conclusions

The study indicated that soil water electrical conductivity measured using in situ collected soil water strongly correlated with HYDRUS 1D prediction. The relative error varied from 5 to 10 % and indicated that the HYDRUS 1D can be used to simulate the salt transport through the root zone.

The application of HYDRUS 1D to examine the long-term impacts of recycled water irrigation indicated that the salt accumulation in root zone showed a cyclical

Table 3 Predicted maximum root zone (0–0.4 m) EC_{SW} during the treatment strategy for different simulated years

Year	100 % RW conc. (EC=0.81 dS/m)			Blended RW conc. (EC=0.42 dS/m)
	Maximum root zone EC_{SW} (dS/m)	Minimum root zone EC_{SW} (dS/m)	Maximum EC_{SW} exceeding maximum threshold of 11.2 dS/m (%)	Maximum root zone EC_{SW} (dS/m)
Year 1	12.72	1.22	14	6.89
Year 2	16.41	3.71	47	8.51
Year 3	17.77	4.44	59	9.11
Year 4	18.26	4.71	63	9.33
Year 5	18.30	4.81	63	9.33

pattern because of the variations in the rainfall and evapotranspiration. Nevertheless, the root zone EC_{SW} increased with time due to recycled water irrigation. The maximum root zone EC_{SW} exceeded the salinity threshold limit of plant tolerance for the simulated years.

The results also showed that by reducing the salt in the irrigation water by 50 %, the root zone EC_{SW} could be reduced to within the vicinity of salinity tolerance limit for growing pasture. Further, the study indicated that there is an increased risk of groundwater salinity with recycled water irrigation if there is a perched water table (<1 m depth below the surface) in field.

Overall, the study identifies a suitable management option of reducing salt accumulation in the root zone due to recycled water irrigation. The management option of blending recycled water with RO-treated water before using it as irrigation water will help in securing sustainability in agricultural irrigation given the increasing reliance on recycled water. The proposed management option is resilient for advanced countries like Australia because water authorities in different major cities in Australia are currently utilising RO-treated recycled water to maintain river health and urban open space irrigation (Sydney Water 2013b; PMHC 2013). Further research may warrant identifying appropriate way of implementing the management option proposed in this study to reduce the risk for salt accumulation in the root zone with longer term recycled water irrigation during drought periods. Further, it is apparent that while irrigating with recycled water, besides monitoring soil moisture, it is necessary to monitor salt levels in the soil.

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