Modelling of the impact of future climate changes on salt accumulation in paddocks of different soil types due to recycled water irrigation

Muhammad Muhitur Rahman, Dharma Hagare, Basant Maheshwari, Peter Dillon and Golam Kibria

ABSTRACT

Recycled water contains elevated amounts of salt compared with irrigation water originating from surface water sources. As such, recycled water, if used for irrigation over a long period of time may increase the root zone salinity. However, the phenomenon depends on variability of climatic condition and soil characteristics. In this study, a salt transport model, HYDRUS 1D, was used to predict long-term salt accumulation in two paddocks containing loamy sand and loam soil. The paddocks are located within Western Sydney University, Hawkesbury campus. Impact of rainfall on salt accumulation was studied with the data from the Global Climate Model for the years 2021–2040. The long-term (20 years) salt accumulation showed a cyclical pattern because of variation in rainfall and evapotranspiration. It was found that soil water electrical conductivity (ECsw) was 24% higher in loam soil paddock compared with that of loamy sand. Amount of leachate in the loamy sand paddock was 27% more than the amount leached from that of loam, which may pose a salinity risk to the groundwater if there is a perched aquifer in the field at a depth <1 m. Results from this study indicate that salt accumulation depends on soil type which seems to be more pronounced under low rainfall condition.

Key words | Global Climate Model, Hawkesbury Water Reuse Scheme, HYDRUS 1D, soil water electrical conductivity

NOMENCLATURE

C5 | Paddock name
CSIRO | Commonwealth Scientific and Industrial Research Organisation
D21 | Paddock name
EC | Electrical conductivity
GCM | Global Climate Model
HWRS | Hawkesbury Water Reuse Scheme
HYDRUS 1D | Salt transport modelling software
MAE | Mean absolute error
PBIAS | Percent bias
PRE | Percent relative error
RMSE | Root mean square error
STP | Sewage treatment plant
TAFE | Technical and further education
VG | van Genuchten
VWC | Volumetric water content (m³/m³)
Ea | Actual evaporation (mm)
Ep | Potential evaporation (mm)
EC | Electrical conductivity (dS/m)
ECe | Electrical conductivity of saturated extract (dS/m)
ECsw | Electrical conductivity of soil water (dS/m)
ET₀ | Reference or potential evapotranspiration (mm)

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Recycled water use, particularly in irrigation, has emerged as a realistic option out of new sources of water to meet water shortages (Tsagarakis 2005; Paranychianakis et al. 2015). Despite significant benefits, recycled water may deteriorate soil health in terms of increased salinity and sodicity. Although several studies in the past (Xu et al. 2013; Rahman et al. 2015) have highlighted the increase of soil salinity due to recycled water irrigation, the phenomenon depends on variability of climatic condition, soil characteristics and irrigation water management. Long-term modelling shows coarse textured soil resulted in lower electrical conductivity (EC) (in terms of soil water electrical conductivity, EC_{SW}) than the finer textured soil (Isidoro & Grattan 2011). As salts are highly soluble, they infiltrate and accumulate in the deeper layers of the soil. The movement of soil water is affected by soil type (specific to the study area), different hydraulic properties of soil (i.e. water holding capacity, hydraulic conductivity, differences in soil water content), irrigation frequency, and most importantly by rainfall variability.

The climate change models suggest an increase in global temperature and variability in rainfall pattern (Yeo 1999; Park et al. 2010; Jung et al. 2015). In Australia the mean temperature may increase 0.4 to 1.8 °C by 2030, with increase of 0.6 to 1.2 °C in eastern Australia (CSIRO & BoM 2015). There is considerable variation in projected rainfall patterns. The projected annual rainfall changes by 2030 in eastern Australia is −13% to +5%, in northern and southern Australia the rainfall changes by −12% to +3% and −15% to +3%, respectively (CSIRO & BoM 2015). The climate change also affects seasonal pattern. For example by 2030 in northern Australia, the projected rainfall may be reduced during spring but increased during summer–autumn, while in southern Australia rainfall projected to be reduced in winter and spring (Cullen et al. 2009). An increased temperature coupled with reduced rainfall may increase the salt accumulation in the upper layers of soil and hence affect plant growth (Rounsevell et al. 1999; Cullen et al. 2009). As water evaporates from soils or is used by the plants, salts are left behind. This phenomenon along with evaporation increases the concentration of salts in the soil with time, until it influences the amount of water a plant can take up from the soil due to the osmotic effect it creates. In many parts of the world saline water has been used as irrigation water due to the increased scarcity of the fresh water (Yeo 1999; Aragüés et al. 2015; Deb et al. 2015). The combined effect of saline water irrigation and changed climate condition further aggravate the soil salinisation in agricultural paddocks (Aragüés et al. 2015; Deb et al. 2015). So, quantifying the soil response in terms of soil water salinity to climate change is critical for the effective management of recycled water irrigation scheme. However, modelling salinity in the upper layers of soil (i.e. root zone) due to long-term recycled water irrigation in changed climatic condition has been overlooked. The main objective of this paper is to apply a salt transport model (HYDRUS 1D) for predicting the impact of soil type on salt accumulation in the root zone soil, when recycled water irrigation is practiced over 20 years. For this purpose, Global Climate Model (GCM) data was used in the modelling.

### MATERIALS AND METHODS

#### Study area

The study area, the Hawkesbury Water Reuse Scheme, is situated within the Hawkesbury Campus of the Western Sydney University in Richmond NSW, approximately 80 km northwest of Sydney (Figure 1). The Hawkesbury Water Reuse Scheme (HWRS) has been built upon...
partnerships between the University and Sydney Water, Richmond TAFE, Hawkesbury City Council, and Clean Up Australia (Booth et al. 2003). The HWRS receives recycled water from Sydney Water’s Richmond sewage treatment plant (STP), which is first collected in a receiving pond, and then pumped up into the first university storage, the Effluent Turkey Nest dam (capacity 93 ML) (Booth et al. 2003). The recycled water is then used in different paddocks as irrigation water. The EC of the recycled water varied between 0.81 and 0.84 dS/m. The mean values of cations were 95.9 mg/L, 20.6 mg/L, 16.9 mg/L and 13.7 mg/L for Na⁺, Mg²⁺, K⁺ and Ca²⁺, respectively. The sodium adsorption ratio was calculated as 3.8.

Soil samples were collected from following two different paddocks. Physicochemical parameters of the collected soil samples were determined in the laboratory and used as input parameters in the HYDRUS 1D model (Šimůnek et al. 2009).

- D21 paddock (S 33°37.478’ E 150°45.706’)
- C5 paddock (S 33°37.199’ E 150°46.182’)

The D21 and C5 paddocks have irrigation history with recycled water since 1989 and 2000, respectively. However, the irrigation in these paddocks was ceased in 2008. This means that D21 and C5 have irrigation histories of approximately 19 years and 8 years, respectively. Soil samples from these paddocks were collected in the months of July–August 2012. Mean annual rainfall in the study area is 800 mm and mean temperature ranges between 10.5 and 23.9 °C.

HYDRUS 1D model and input parameters

The HYDRUS 1D model was used to simulate one dimensional water flow and solute transport in incompressible, porous, variably saturated soil under transient system. A number of commercially developed models are available to simulate solute transport in root zone, which include SALTMED (Ragab 2002), SaltMod (Bahceci & Nacar 2007), SWAP (Kroes et al. 2000), and ENVIRO-GRO (Chen et al. 2010). However, HYDRUS 1D was used in this study because the model serves the purpose of the study. Also, the available input data was well suited for using in HYDRUS 1D. In HYDRUS 1D, 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 advection-dispersion equation for non-reactive solutions (Dag et al. 2015). 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 in Šimůnek et al. (2009).

The HYDRUS 1D model was validated using data from a continuous soil column study conducted in the laboratory. Soil samples collected from D21 and C5 paddocks were re-packed in columns to form a soil profile with a height of 0.3 m. Three columns from each soil type were prepared. A soil water sampler at a depth of 0.2 m from the soil surface was installed in one of the three columns. EC of collected soil water was measured and is denoted as ECSW. All three columns of D21 paddock soil were packed in such a way that the same bulk density of 1,460 kg/m³ was maintained. In the case of C5 paddock soil columns, packing bulk density of 1,370 kg/m³ were maintained. The soil column experiment was conducted over a period of 187 d. Irrigation (recycled) water was applied three times per week; total amount of 790 mm of recycled water was applied during the study period. Soil water samples for measuring ECSW were collected at the end of each week.

The validation work was carried out for a period of 187 d for both types of soil. Additionally, the model was run for spin-up period for 50 d and 100 d for D21 and C5 soil, respectively. The spin-up period is provided so that the model will be adequately equilibrated and have a reasonably realistic soil condition (in terms of volumetric water content (VWC) and ECSW) at the beginning of the simulation period. Some of the input parameters are summarised in Table 1. The physicochemical parameters (Table 1) such as soil type, water flow parameters (van Genuchten 1980), VWC and electrical conductivity of saturated extract (ECe) for D21 and C5 paddock soils were determined in the laboratory.

The validated HYDRUS 1D model, by column study, was used to predict salt accumulation in D21 and C5 paddocks.

### Table 1 | Input parameters of HYDRUS 1D model used in the validation study

<table>
<thead>
<tr>
<th>Description</th>
<th>D21 paddock</th>
<th>C5 paddock</th>
</tr>
</thead>
<tbody>
<tr>
<td>Depth of soil below the soil surface (m)</td>
<td>0.3</td>
<td>0.3</td>
</tr>
<tr>
<td>Simulation period (d)</td>
<td>187</td>
<td>187</td>
</tr>
<tr>
<td>Hydraulic model</td>
<td>VG-Mualem</td>
<td>VG-Mualem</td>
</tr>
<tr>
<td>Soil type</td>
<td>Loamy sand (NCST 2009): sand = 88.1%, silt = 6.0%, clay = 5.9%</td>
<td>Loam: sand = 67.6%, silt = 18.0%, clay = 14.4%</td>
</tr>
<tr>
<td>Bulk density (kg/m³)</td>
<td>1,460</td>
<td>1,370</td>
</tr>
<tr>
<td>Water flow parameters</td>
<td>( \theta_r = 0.03 \text{ m}^3/\text{m}^3, \theta_s = 0.41 \text{ m}^3/\text{m}^3, \alpha = 0.006, n = 2.771, K_s = 298.26 \text{ cm/d} )</td>
<td>( \theta_r = 0.05 \text{ m}^3/\text{m}^3, \theta_s = 0.44 \text{ m}^3/\text{m}^3, \alpha = 0.005, n = 2.314, K_s = 74.35 \text{ cm/d} )</td>
</tr>
<tr>
<td>Longitudinal dispersivity (cm⁻¹)</td>
<td>1.3 (Vanderborght &amp; Vereecken 2007)</td>
<td>3.9 (Vanderborght &amp; Vereecken 2007)</td>
</tr>
<tr>
<td>Initial condition</td>
<td>VWC = 0.09 m³/m³, ECSW (dS/m) = 2⁺ECₑ (Ayers &amp; Westcot 1985), ECₑ = 0.375 dS/m</td>
<td>VWC = 0.18 m³/m³, ECSW (dS/m) = 2⁺ECₑ (Ayers &amp; Westcot 1985), ECₑ = 0.340 dS/m</td>
</tr>
<tr>
<td>Water flow boundary condition</td>
<td>Upper BC: atmospheric with surface layer Lower BC: free drainage</td>
<td></td>
</tr>
<tr>
<td>Solute transport boundary condition</td>
<td>Upper BC: concentration flux Lower BC: zero gradient concentration</td>
<td></td>
</tr>
<tr>
<td>Type of solute transport model</td>
<td>Equilibrium model</td>
<td></td>
</tr>
<tr>
<td>Molecular diffusion coefficient (cm²/d)</td>
<td>1.75 (James &amp; Rubin 1986)</td>
<td></td>
</tr>
<tr>
<td>Irrigation (recycled) water EC (dS/m)</td>
<td>0.83</td>
<td></td>
</tr>
<tr>
<td>Meteorological parameters</td>
<td>Recorded in the laboratory</td>
<td></td>
</tr>
</tbody>
</table>
paddocks under field conditions. The prediction of EC_{SW} was conducted over a period of 20 years (from 2021 to 2040) with a spin-up period of three years. For the spin-up period, climate data for 2021 to 2023 was used. The 20 years period from 2021 to 2040 is considered in this study as that was only the data set which was available to the research team. For the prediction under field conditions, 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 validation of the model (Table 1). For the field prediction, a soil profile up to 1 m below the ground level was considered. Bulk densities and longitudinal dispersivities were 1,500 kg/m^3 and 1,470 kg/m^3 (measured in the field), and 20 cm^{-1} and 21.7 cm^{-1} (Vanderborght & Vereecken 2007) for D21 and C5 paddocks, respectively.

The daily rainfall and evaporation data of future (2021 to 2040) climate condition was collected from Sydney Catchment Authority for the weather station (Station number 067021) at Hawkesbury campus, Western Sydney University. These future climate data are downscaled data from CSIRO Mark 3.0 GCM using statistical downscaling method to a finer spatial scale (about 5 × 5 km) to be used by the forecasting model (Haque et al. 2014). Three future climatic scenarios were proposed by Intergovernmental Panel on Climate Change corresponding to three different greenhouse gas emission conditions: B1 (low emission); A1B (medium emission); and A2 (high emission) (IPCC 2000). In this study, data from low emission scenario is investigated. It should be noted that the purpose of this study is to quantify salt accumulation using future climatic conditions, rather than investigating the impact of greenhouse gas emission scenario on salt accumulation. Therefore, use of any emission scenario is sufficient to serve the purpose of this study. As such, low emission scenario was selected. Under low emission scenario, two future climatic scenarios, i.e. minimum and maximum rainfall (based on 20-year total) were selected. In this paper, they are termed as low rainfall and high rainfall scenarios. From these two scenarios, it is possible to obtain a clear picture of minimum and maximum impact of rainfall on salt accumulation.

An irrigation schedule was calculated based on Smith et al. (2012) for rye pasture by taking the daily rainfall into account. The soil moisture deficit was calculated based on readily available water (share of the difference between field capacity and permanent wilting point, which is above the soil moisture limit without leading to water stress) in the root zone. The maximum amount of irrigation water to be applied per irrigation was calculated based on the product of the average water holding capacity of the soil and root depth of rye pasture. For loamy sand (D21) and loam (C5) soil, water holding capacity was 55 mm/m (SARDI 2014) and 80 mm/m (Allan et al. 1997), respectively. The average root depth of rye pasture was assumed as 0.35 m (Allan et al. 1997). For each irrigation event, 19 and 28 mm of irrigation water was used for D21 and C5 paddock soils, respectively. A crop factor of pasture was used for different months varying between 0.4 and 0.7 (Allan et al. 1997). Using Smith et al. (2012) method, total number of irrigation events per year varied with the rainfall, which is shown in Table 2 for D21 and C5 paddocks. Variation of rainfall and potential evapotranspiration (ET_{0}) over the study period is shown in Figure 2.

In HYDRUS 1D, potential evaporation (E_{p}) and potential transpiration (T_{p}) are required as input data. The future pan evaporation data (obtained from GCM) was converted to ET_{0} by multiplying a site-specific factor of 0.8. This value was earlier used by Yiasoumi et al. (2008). ET_{0} was then divided into potential transpiration (T_{p}) and potential evaporation (E_{p}) using Beer’s Law (Wang et al. 2009), which was then entered into HYDRUS 1D. The T_{p} and E_{p} varied in the range of 0.1 to 11 mm/d and 0.02 to 3 mm/d, respectively. The model then converts them into actual evaporation (E_{a}) and transpiration (T_{a}) based on the available soil moisture content. 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.

In addition to the visual comparison, four statistical parameters, namely, mean absolute error (MAE), root mean square error (RMSE), percent relative error (PRE) and percent bias (PBIAS), were used to evaluate the goodness of fit between measured and predicted data. An ideal value of MAE, RMSE, PRE and PBIAS is zero. Low values of PBIAS indicate better simulation results by the model.
### Table 2: Irrigation scheduling based on soil moisture deficit in the root zone

<table>
<thead>
<tr>
<th>Year</th>
<th>Low rainfall scenario (mm)</th>
<th>High rainfall scenario (mm)</th>
<th>Yearly total rainfall in past years</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>D21 paddock</td>
<td>CS paddock</td>
<td>Year</td>
</tr>
<tr>
<td>2021</td>
<td>660</td>
<td>758</td>
<td>2095</td>
</tr>
<tr>
<td>2022</td>
<td>418</td>
<td>1,061</td>
<td>2096</td>
</tr>
<tr>
<td>2023</td>
<td>451</td>
<td>1,215</td>
<td>1997</td>
</tr>
<tr>
<td>2024</td>
<td>676</td>
<td>683</td>
<td>1998</td>
</tr>
<tr>
<td>2025</td>
<td>474</td>
<td>753</td>
<td>1999</td>
</tr>
<tr>
<td>2026</td>
<td>745</td>
<td>828</td>
<td>2000</td>
</tr>
<tr>
<td>2027</td>
<td>685</td>
<td>864</td>
<td>2001</td>
</tr>
<tr>
<td>2028</td>
<td>916</td>
<td>645</td>
<td>2002</td>
</tr>
<tr>
<td>2029</td>
<td>617</td>
<td>1,113</td>
<td>2003</td>
</tr>
<tr>
<td>2030</td>
<td>563</td>
<td>675</td>
<td>2004</td>
</tr>
<tr>
<td>2031</td>
<td>906</td>
<td>756</td>
<td>2005</td>
</tr>
<tr>
<td>2032</td>
<td>574</td>
<td>819</td>
<td>2006</td>
</tr>
<tr>
<td>2033</td>
<td>582</td>
<td>703</td>
<td>2007</td>
</tr>
<tr>
<td>2034</td>
<td>696</td>
<td>1,370</td>
<td>2008</td>
</tr>
<tr>
<td>2035</td>
<td>554</td>
<td>1,015</td>
<td>2009</td>
</tr>
<tr>
<td>2036</td>
<td>1,248</td>
<td>987</td>
<td>2010</td>
</tr>
<tr>
<td>2037</td>
<td>808</td>
<td>1,071</td>
<td>2011</td>
</tr>
<tr>
<td>2038</td>
<td>643</td>
<td>514</td>
<td>2012</td>
</tr>
<tr>
<td>2039</td>
<td>914</td>
<td>1,117</td>
<td>2013</td>
</tr>
<tr>
<td>2040</td>
<td>901</td>
<td>904</td>
<td>2014</td>
</tr>
</tbody>
</table>

**Figure 2**: Variation of rainfall and ET0 over the study period.
where positive and negative values represent underestimation and overestimation bias, respectively, in the simulated results.

RESULTS AND DISCUSSION

Validation of HYDRUS 1D model with column study result

The observed EC\textsubscript{SW} and the one predicted by HYDRUS 1D over 187 d are shown in Figure 3 for the depth of 0.2 m (average root zone depth) for D21 and C5 soil on all sampling occasions. As can be seen in the figure, HYDRUS 1D was able to predict the general trend in the salt accumulation. However, there are some variations between the predicted and observed EC\textsubscript{SW} values.

The calculated MAE, RMSE, RE and PBIAS between observed and simulated EC\textsubscript{SW} are shown in Table 3. The RMSE showed almost similar prediction capability of HYDRUS 1D for both types of soil and agreed with the range reported by other researchers (0.21 to 3.73) who used HYDRUS 1D for salt transport modelling (Forkutsa et al. 2013; Ramos et al. 2014; Kanzari et al. 2015). For D21 soil, HYDRUS 1D overestimated the observed EC\textsubscript{SW} and for C5 soil, it underestimated the observed EC\textsubscript{SW} by about 4.5%.

Agreement between simulated and observed results on EC\textsubscript{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\textsubscript{SW} (in terms of different goodness of fit indices) might be because of the preferential flow (due to the existence of cracks, roots, and some gaps between soil and column material) and locally entrapped air (Peck 1969; Phillips 2006), which is not considered by this study. However, the existence of preferential flow in column studies is reported by different researchers (Camobreco et al. 1996; Duan et al. 2011).

Rainfall amount and risk of salt accumulation in field condition

Results obtained from the simulation of 20 years of recycled water irrigation for low rainfall scenario are presented in Figure 4 for D21 and C5 paddocks. The reported salinity represents average root zone (from surface to 0.4 m) EC\textsubscript{SW} over the entire year. As expected, the root zone salinity shows a cyclical pattern (alternatively increasing and decreasing with time). The cyclical pattern of the salt accumulation in the soil profile is linked to the variation of rainfall and evapotranspiration (Rahman et al. 2015).

![Figure 3](https://iwaponline.com/ws/article-pdf/16/3/653/412346/ws016030653.pdf)

**Figure 3** Observed and simulated soil water concentration (EC\textsubscript{SW}) at 0.2 m depth during different sampling times: (a) D21 soil; and (b) C5 soil.

<table>
<thead>
<tr>
<th>Goodness of fit indices</th>
<th>D21 Observed EC\textsubscript{SW} vs. simulated EC\textsubscript{SW}</th>
<th>C5 Observed EC\textsubscript{SW} vs. simulated EC\textsubscript{SW}</th>
</tr>
</thead>
<tbody>
<tr>
<td>RMSE (dS/m)</td>
<td>0.20</td>
<td>0.20</td>
</tr>
<tr>
<td>MAE (dS/m)</td>
<td>0.17</td>
<td>0.16</td>
</tr>
<tr>
<td>PRE (%)</td>
<td>10.78</td>
<td>5.40</td>
</tr>
<tr>
<td>PBIAS (%)</td>
<td>-4.49</td>
<td>4.35</td>
</tr>
</tbody>
</table>

\[
\text{MAE} = \frac{1}{N} \sum_{i=1}^{N} |O_i - P_i|; \quad \text{RMSE} = \sqrt{\frac{\sum_{i=1}^{N} (O_i - P_i)^2}{N-1}}; \quad \text{PRE} = 100 \frac{\sum_{i=1}^{N} |O_i - P_i|}{\sum_{i=1}^{N} |O_i|}; \quad \text{PBIAS} = \frac{\sum_{i=1}^{N} (O_i - P_i)^2}{\sum_{i=1}^{N} |O_i|} \times 100.
\]

where, \( O_i \) represents observed values; \( P_i \) represents predicted values; and \( N \) represents the number of observations.
In Figure 4 it is evident that salt accumulation in the root zone varies with rainfall. Higher yearly rainfall amount resulted in lower salt accumulation. This is also reported by other researchers, such as Selle et al. (2010). However, the change of salt accumulation from one year to the next is non-linear to the amount of change in rainfall. For example, for D21 paddock (Figure 4(a)), from year 2034 to 2035, rainfall was decreased by 20% and the salt accumulation increased by 90% (from 3.1 to 5.8 dS/m). From 2035 to 2036, rainfall increased by 125% which reduced the salt accumulation by about 69% (from 5.8 to 1.8 dS/m). A similar pattern was observed for C5 paddock (Figure 4(b)), where salt accumulation increased by 60% (from 4.2 to 6.7 dS/m, from year 2034 to 2035) and decreased by about 70% (from 6.7 to 2.0 dS/m, from year 2035 to 2036) for a corresponding decrease and increase of rainfall of 20 and 125%, respectively. However, lowering of soil salinity is not only dependent on the amount of rainfall, but also on rainfall intensity and distribution over time affecting the infiltration.

**Salt accumulation under low and high rainfall scenarios**

Figure 4 shows the dependence of root zone EC_{sw} on variation of rainfall, however to get a clear range of variation of salt accumulation, impact of low and high rainfall scenarios...
on root zone salt accumulation need to be investigated. The results of the simulation for D21 and C5 paddocks are shown in Figure 5. There are significant differences among the amount of total yearly rainfall under the low and high rainfall scenarios (Table 2). This caused low salt accumulation in high rainfall scenario than under low rainfall scenario (Figure 5). In most of the simulation years in the study period annual ECSW was lower in high rainfall scenario compared with the low rainfall scenario. However, during some years (such as 2028 and 2031) higher salt accumulation occurred in the high rainfall scenario. This is due to the occurrence of high rainfall in low rainfall scenario during those particular years (Table 2). Overall, there is similarity in the trend of salt accumulation in both types of soil under both scenarios.

The highest difference in the ECSW between high and low rainfall scenarios were in 2023 (Figure 5). In this year, total annual rainfall under high rainfall scenario was 2.7 times higher than the low rainfall scenario (451 mm in low rainfall scenario and 1,215 mm in high rainfall scenario). Reduction in rainfall caused 6.6 times increase (from 1.6 to 10.5 dS/m) in ECSW in 2023 for D21 paddock and 9.6 times increase (from 1.4 to 13.5 dS/m) for C5 paddock in the same year. The results indicate that for the GCM projected rainfall condition (for both low and high rainfall scenarios) from the year 2021 to 2040, salt accumulation in D21 paddock varied between 1.1 and 10.5 dS/m. On the other hand, for C5 paddock, the salt accumulation varied from 1.0 to 13.5 dS/m. The reasons for the above difference in salt accumulation are explained in the next section.

**Impact of soil type on salt accumulation**

Given that the higher amount of salt was accumulated during the low rainfall scenario, it would be justified to use this scenario to compare the maximum impact of rainfall on salt accumulation for two types of soil of the paddocks (i.e. loamy sand of D21 and loam of C5 paddocks). Figure 6 shows the salt accumulation in D21 and C5 soils under the low rainfall conditions. Over the

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*Figure 5* | Effect of yearly total rainfall on annual average root zone ECSW of two paddocks under low and high rainfall scenarios: (a) D21 paddock; and (b) C5 paddock.
simulation period, in most of the simulation years, there was a higher amount of salt accumulated in C5 paddock compared with the D21 paddock. Maximum salinity in C5 paddock was simulated as 13.5 dS/m, which was 28% more than that of D21 paddock; both occurred in the year 2023. Among other years, in 2021, salt accumulation in C5 paddock was 7.6 dS/m, which was 53% more than that of D21 ECSW; and in the year 2037, salt accumulation in C5 paddock was 2.2 dS/m, which was about 1% more than that of D21 ECSW. When averaged over 20 years, salt accumulated 24% more in C5 paddock compared with D21 paddock.

From the above discussion, it is clear that soil type plays an important role in salt accumulation in the root zone. Soil texture is associated with its capability to infiltrate water, to hold available water (i.e., water holding capacity), and the its ability to exchange ions (Hillel 1980). Clay-type soils generally comprise of more cation exchange sites owing to their relatively higher surface area (Hillel 1980), and therefore able to accumulate salt or more specifically accept excess sodium which contribute to the increase of soil water salinity. A coarse texture soil, on the other hand, has less total surface area, fewer exchange sites, and salt is easily transported downward through its larger particle size (Hillel 1980; Wang et al. 2009). Therefore, the variation of salt accumulation in C5 and D21 paddock soil can be attributed to the clay content of respective soil. As shown in Figure 6, the salt accumulation within the soil which contains more clay (C5 soil) appears to be higher compared with the soil which contains more sand (D21 soil). The C5 soil used in this study contains about 41% more clay content than that of D21 soil (Table 1).

The variation of salt accumulation in the root zone for both types of soil in a given year is also evident, which is represented using error bars in Figure 6. These error bars represent maximum and minimum root zone EC_{SW} which occurred in each year. The maximum EC_{SW} shown for each year are sustained over a short duration of time (1 d for D21 soil and 1–3 d for C5 soil). The reason for the longer duration of EC_{SW} in C5 soil may link to the higher clay content and lower hydraulic conductivity of C5 soil (Table 1) compared with D21 soil. According to Hernández & Almansa (2002), short-term salt stress (up to 48 hours) may decrease the osmotic potential in plants and reduce

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**Figure 6** | Salt accumulations in D21 paddock compared with C5 paddock under low rainfall condition. The error bar indicates the minimum and maximum root zone EC_{SW} occurring over a year.
the amount of water in the leaf. Growth of leaves may also be interrupted. However, both growth and water content in leaves may be restored within a period of 8–24 hours (Yeo et al. 1997; Hernández & Almansa 2002). As such, long-term exposure to salinity (i.e. weeks to years) may affect the photosynthetic area, eventually limiting the growth of leaves (Munns & Termaat 1986).

Impact of soil type on leaching

Over the simulation period, amount of leaching in D21 paddock was more than that of C5 paddock. On an average, annual total amount of leachate from D21 paddock was 162 mm with a range of 25 to 449 mm (Figure 7). This wide range of leaching is because of the variation of yearly rainfall. Rainfall of adequate intensity and duration may flush the soil water and salts contained in it from upper to bottom layers of soil profile (Selle et al. 2010). As discussed in the previous section, clay-type soil has more water holding capacity and less capability to drain water because of its smaller pore diameters. On the other hand, the water holding capacity of coarse texture soil is lower and water drains fast because of its larger pore diameters. Consequently, leaching occurs more in coarse textured soil than clay-type soil. Similar observations were made from the simulation in current study for the soil of D21 (loamy sand) and C5 (loam) paddocks (Figure 7). Compared with the D21 paddock, C5 paddock produced about 21% less amount of leachate. The study of the amount of leachate from paddocks is important especially, if it includes underlying shallow aquifer and if no drainage system is in place to move out the leachate from the paddocks. This is because the washed out salt from the root zone may end up contaminating groundwater aquifers. From Figure 7 it is evident that salt was leached more than overall average in some simulation years and this is more predominant in D21 paddock. As shown in the figure, in the years 2036 and 2040, where total yearly leached amount from both the paddocks was two to three times more than the respective overall average leached amount. Beveridge (2006) monitored the groundwater table at D21 paddock site from January 2004 to April 2005 and reported that the depth of the water table of the perched aquifer was between 1.4 to 2.4 m. It should be mentioned that there is no underlying drainage system in D21 and C5 paddocks. Therefore, continuous irrigation using recycled water over a long period of time may impact on the salinity levels of the groundwater in the perched aquifer.

Seasonal variation of root zone salt accumulation

The D21 and C5 paddocks are situated in the temperate climatic zone of Australia, where the seasons are divided into four groups: 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. In this section, variability of salt accumulation in different seasons is explained.

The seasonal variation of root zone $EC_{SW}$ in D21 and C5 paddocks is shown in Figure 8. $EC_{SW}$ values plotted in Figure 8 are the average values over the season. Comparison among seasonal salt accumulation was conducted for the year 2023. This particular year was selected because of its highest yearly $EC_{SW}$ over the simulation period (Figure 6). The root zone $EC_{SW}$ in both D21 and C5 paddocks was found to be higher than salinity tolerance thresholds for pastures (in terms of $EC_{SW}$) including clover and rye. According to NRMMC-EPHC-AMC (2006) the threshold varies from 3.0 to 5.0 dS/m. The root zone $EC_{SW}$ of C5 paddock was found more than $EC_{SW}$ of D21 paddock for all four seasons. As shown in Figure 8, the winter season showed the highest amount of salt accumulation compared with other seasons, which is because the winter season had lowest rainfall in the year 2023. Summer, autumn and spring seasons of the year 2023 had 2.4, 1.8 and 2.8 times, respectively, more rainfall than winter season in this year. The $EC_{SW}$ of C5 paddock in summer, autumn, winter and spring season exceeds the maximum salinity threshold limit by 2.2, 2.4, 3.3 and 2.9 times, respectively. In D21 paddock, root zone $EC_{SW}$ exceeds the threshold by 2.2, 1.9, 2.2 and 2.1 times in summer, autumn, winter and spring seasons, respectively. The exceedance of root zone $EC_{SW}$ of the maximum threshold limit salinity tolerance in both the paddocks may lead to the detrimental circumstances to the yield of pastures irrigated with recycled water in these paddocks.

Generally, the findings of this study, such as simulated salt accumulation in two different types of soil, complies with the mechanism of salt accumulation, leaching, and seasonal variation of $EC_{SW}$ reported in the literature. The significance of this paper, however, lies in explaining the impact of salinity due to recycled water irrigation under changing climate conditions, particularly under low rainfall scenarios. According to Rounsevell et al. (1999), farmers will need to reconsider their management options for the adaptation of changed climate condition, which include different crop selection and changes in irrigation practice. The findings from this study can support farmers in conceiving irrigation strategies to cope with the impact of climate change on soil salinity.

**CONCLUSIONS**

It is recognised from the discussion and salt transport modelling results that long-term irrigation with recycled water may progressively increase the salt accumulation in the root zone in D21 and C5 paddocks. This necessitates an examination of possible management options which can relieve the salt accumulation due to long-term irrigation using recycled water. One of the management options includes reducing salt level in the recycled water before using as irrigation water. This can be achieved by blending or mixing recycled water with fresh water. Sources of fresh water may be of reverse osmosis (RO) treated recycled water and stormwater. RO produce good quality of water in terms of salinity, however, the process is expensive. Instead, it is possible to blend harvested stormwater with untreated recycled water to reduce the salinity in irrigation water. Stormwater is expected to contain relatively lower salt levels ($EC$ of 0.17 to 0.34 dS/m) as compared with the recycled 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. In addition, conventional management options such as leaching of salt may be used to reduce the salt build-up in the soil profile.

Overall, the long-term (20 years) modelling with future climate data presented in this paper indicates that the salt accumulation in the soil depends on the soil type, which
particularly appears to be important under low rainfall conditions. The approach presented in this paper can be used to select an appropriate method for irrigation using recycled water that not only satisfies the maintenance of desired soil moisture levels, but also ensures that the soil water salinity at the root zone does not exceed the recommended threshold levels.

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