Sustainable expansion of irrigated agriculture and horticulture in Northern Adelaide Corridor: Task 2 - Modelling nutrient and chemical fate to support the long-term sustainability of the use of recycled water

APPENDICES 8 to 12

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Appendix 9: Management of irrigation associated hazards for greenhouse crops

Contents

Executive summary .......................................................................................................................... 327
1. Introduction ............................................................................................................................ 329
2. Materials and methods .......................................................................................................... 332
3. Results and Discussion ......................................................................................................... 335
4. Conclusions ............................................................................................................................. 342
References ...................................................................................................................................... 343
Executive summary

Long-term evaluation of irrigation induced transformations in the soil is essential for optimal water management and devising effective irrigation scheduling for crops, including protected crops. In this study, the multi-component numerical model HYDRUS-1D UNSATCHEM was used to assess the effects of long-term (2018-2050) irrigation on salt build-up in soil under greenhouse conditions. Blended water (recycled water and harvested rainwater) was used to irrigate soil grown greenhouse vegetables (tomato, cucumber, capsicum, and eggplant). Simulations provided insight into the development of irrigation induced chemical transformations in the soils and which management options provide for a sustainable use of irrigation water. The management scenarios include 4 leaching fractions (LF), i.e. LF0, LF0.15, LF0.2, and LF0.3, accounting for 0, 15, 20, and 30% excess water applied, respectively. We also consider four annual levels of gypsum application, i.e. 0 (G0), 10 (G10), 15 (G15) and 20 (G20) meq/kg soil, respectively. The model simulated annual root water uptake by cucumber, tomato, capsicum and eggplant was 303, 476, 642 and 649 mm, respectively, in response to a temperature based irrigation schedule that did not account for a leaching fraction. Whereas, annual drainage component accounted for 4.1-6.1% of the amount of irrigation to different crops. On the other hand, the average salinity in the soil solution (ECsw) at the end of the simulation (year 2050) increased to 6.5, 7.6, 8.7 and 9.3 dS/m, respectively, for cucumber, tomato, capsicum and eggplant. Results further indicate that salinity continues to build up in the soil at a nearly constant rate. Similarly, the ESP values at the end of the simulation were 30.8, 27.1, 33.2, and 31.4 % for tomato, capsicum, eggplant and cucumber crops, respectively. Based on initial ESP values in the range 19-25, this increase indicates a high sodicity development in the soil. Results show that temperature based irrigation scheduling for greenhouse crops (with a zero leaching fraction) could lead to the emergence of salinity and sodicity hazards in the soil. Analysis of further management scenarios suggested that a 15-20% (LF0.15- LF0.2) higher irrigation amount coupled with an annual gypsum addition of 10 meq/kg soil (G10, 1.7 t/ha), kept the salinity and sodicity (SAR, ESP) below critical thresholds. This scenario created a favourable soil environment for long-term sustainable vegetable production under glasshouse condition. It reduced the soil pH by 7.5-8 % and maintaining this within an adequate range (7.74-7.8), allowing for optimal plant nutrient availability. Overall, our modelling results indicate that inappropriate long-term irrigation applications can significantly impact soil health. Therefore, adoption of informed irrigation scheduling for the
long-term use of blended water is essential for sustainable glasshouse vegetable production. Apart from salinity, other constraints such as high ESP of the soils needs to be adequately addressed by applying the proper amount of soil amendments.
1. Introduction

Protected agriculture is a steadily growing sector all over the world (FAO, 2017), with greenhouse vegetable production as one of the most important and profitable agricultural systems. Greenhouses essentially provide an opportunity to partially modify the climatic parameters for developing an effective growing system under otherwise adverse conditions, including climate variability and change. In Australia, the protected cropping industry had an estimated value of $1.8 billion at the farm-gate, which accounts for approximately 20% of the total gross value of production of vegetables and cut flowers (PCA, 2019). Main crops grown in greenhouses in Australia are tomato, capsicum, eggplant, cucumber and melons, with tomatoes accounting for about half the area. Nevertheless, most greenhouses (up to 80%) in Australia are low cost structures with soil grown crops (Parks et al., 2009), though the number of high technology structures on soilless growing media are increasing in recent years to meet specific market requirements.

Water availability and its quality are the most important factors that determine the expansion of protected agriculture. It is well understood that greenhouse crops cannot be grown profitably with highly saline water (Stanghellini et al., 2007). To cope with these problems, many irrigators have to make use of several water sources, which are either conventional (surface and groundwater), or non-conventional (recycled waste water, desalinated water, brackish water, etc.). The heterogeneous quality of different water sources adds a higher level of complexity to irrigation water management and their use poses a new challenge to managers and irrigators (Reca et al., 2018). Therefore, in many arid and semiarid regions, protected horticulture is increasingly facing the salinization of soil and water resources (Pardossi et al., 2004). On the other hand, most of the crops including vegetables grown in the greenhouses are sensitive to water and salinity; therefore, the knowledge of crop response to salinity is important for optimal crop management (Stanghellini et al., 2007). Similarly, in the Northern Adelaide Plains (NAP) region of South Australia, yields of soil grown tomato and cucumber crops had been steadily declining for several years due to high salinity (high sodium and chloride) levels in the soil (Ferguson et al., 2012). The reason for the decline was found to be poor quality of the primary water source, which was either saline bore or recycled water with an EC of 2-4 dS/m. Due to limited storage capacity, only a very limited amount of rain water was captured from greenhouse roofs that could be blended to dilute the salinity of irrigation water. Likewise, Shi et al. (2009) observed high nutrient accumulation which leads to higher
soil salinity in long-term greenhouse cultivation. Some nutrients that accumulate in the root zone may leach downward with the application of irrigation water, possibly causing soil and groundwater contamination (Kim et al., 2008). On the other hand, most greenhouse farmers are following irrigation and fertigation practices based on their experiences (Thompson et al., 2007), without monitoring or controlling the soil water, nutrient and salt status. Therefore, accurate estimations of crop water requirement are needed to avoid, on the one hand, salt build up in the soil profile, and on the other hand, excess water application that would negatively impacts on nutrient availability for plants and leaching of agrochemicals to groundwater (Blanco and Folegatti, 2004).

Irrigation application decisions within a closed growing structure depends on the accurate estimation of the crop evapotranspiration (ETc) for effective irrigation scheduling. However, direct measurement methods for crop water requirement (e.g., weighing lysimeters, stem gauges) are either expansive or abrasive and may not be practical to implement in commercial greenhouse settings (Villarreal-Guerrero et al., 2012). Alternatively, simulation models offer cost effective and non-destructive techniques for crop water use. Among the models, the FAO-56 methodology (Allen et al., 1998) has been used worldwide to determine crop water requirements under open field conditions and this methodology can also be applied to greenhouse crops (Bonachela et al. 2006; Qiu et al., 2015). In order to use the FAO-56 methodology for greenhouse crops, measurement of greenhouse reference evapotranspiration (ET0) and crop coefficients (Kc) are required (Fernández et al., 2010). Möller and Assouline (2007) observed that the daily ET0 under a cropped screen house, calculated following the FAO-56 methodology, averaged 62% of the estimated outdoor value. The main reason for the reduction in ET0, compared to outdoors, was the significant reduction in solar radiation inside the greenhouse (56% of open field conditions), while the lower wind speeds inside contributed to a smaller extent. Farias et al. (1994) observed that the reference evapotranspiration (ET0) inside greenhouses was always lower than outdoors, ranging on 45 to 77% of that verified outside. Braga and Klar (2000) observed that the values of ET0 were 85 and 80% of the outdoor ET0 for greenhouses oriented east/west and north/south, respectively. However, Harmanto et al. (2005) found that the crop evapotranspiration (ETC), estimated based on the measured climate data within the greenhouse, matched the 75–80% of the ETC computed with the climatic parameters observed in the open environment. Several software have been developed that are based on the FAO-56 method and use a single Kc approach, such as PrHo V2.0
Due to the complexity involved in measuring the climate variables and crop data, such as leaf area index, other simple methods have been explored such as the temperature-based Hargreaves equation for accurately estimating greenhouse ET0 (Fernández et al., 2010). Empirical approaches widely used under open field conditions that only require temperature and greenhouse radiation data (Allen et al., 1998; Itenfisu et al., 2003; Gavilán et al., 2006), could be easily applied for determining greenhouse ET0 (Boulard and Wang, 2000). Nikolaou et al. (2017) found a good correlation between leaf temperature and transpiration for soilless greenhouse cucumber crop, with the prediction equation being validated under different greenhouse climatic conditions.

However, irrigation practices on the ground have often oversimplified the complex nature of the interacting climate, soil and crop variables involved in the estimation of crop water requirement. For example, outdoor temperature based irrigation scheduling is practiced by the growers for irrigating greenhouse crops in the Northern Adelaide Plains by considering only 60% of outdoor ETC (Awad et al., 2019). While this simplified approach requires less data, there is need to test its effectiveness on a long-term basis. Essentially, observations of soil health over one or several seasons are unable to reveal the potential adverse impacts of irrigation induced problems such as salinity and sodicity, which usually arises only after many years of adopting a particular irrigation schedule.

Therefore, in the present study we evaluate the impact of long-term use of a temperature based irrigation schedule to soil grown tomato, cucumber, capsicum and eggplant under greenhouse conditions. We explore management options such as increased leaching fractions and annual gypsum applications, and seek to identify how to make best use of blended water for sustainable irrigation in the NAP region. To achieve these objectives, we used the multi-component major ion chemistry module UNSATCHEM of the HYDRUS-1D numerical model (Šimůnek et al., 2016). We focus our simulations on understanding the long-term irrigation induced salinity and sodicity risks to soil and shallow groundwater, and assess the efficacy of practical management options, including gypsum amendments and increased leaching. In this study, we are also implementing a new capability in the UNSATCHEM module, i.e. applying a solid form of gypsum annually for reducing the ESP levels in the soils. The outcomes of these
long-term simulations, accounting for the future climate, would help in devising better irrigation guidelines for growing vegetable crops under protected environments while also coping with the adverse impact of climate change.

2. Materials and methods

The study focussed on conducting long-term (1970-2050) simulations using the HYDRUS-1D simulator with the UNSATCHEM module (Šimůnek et al., 2016) for evaluating the impact of long-term use of blended water for irrigation of crops grown under low-cost greenhouse conditions in the NAP region of South Australia. UNSATCHEM is a multi-component major ion chemistry process based numerical model which requires numerous input data for soil, crop, water and climate conditions of the study site. Therefore, soil samples from two greenhouses were collected from the NAP region (Awad et al., 2019) from 3 locations at two depths (0-20 and 20-30 cm). These samples were analysed for their physico-chemical properties (Oliver et al., 2019) to generate soil hydraulic and chemical properties for the UNSATCHEM model. The average van Genuchten-Mualem (van Genuchten, 1980) soil hydraulic parameters for two locations are given in Table 1 and soil exchange properties are shown in Table 2. Although a 200-cm-deep domain was adopted for the simulations, soil hydraulic parameters obtained in 20-30 cm layer of the greenhouse soils were extended to the bottom of the 200-cm-deep domain.

Table 1 Soil hydraulic properties of two soils (S1 and S2) for two depths (0-20, 20-30 cm) used in the modelling simulations (this report, Appendix 5).

<table>
<thead>
<tr>
<th>Soil</th>
<th>Depth (cm)</th>
<th>$\Theta_r$ (cm$^3$/cm$^3$)</th>
<th>$\Theta_s$ (cm$^3$/cm$^3$)</th>
<th>$\alpha$ (cm$^{-1}$)</th>
<th>n (-)</th>
<th>$K_s$ (cm/d)</th>
<th>l (-)</th>
<th>Bulk density (g/cm$^3$)</th>
</tr>
</thead>
<tbody>
<tr>
<td>S1</td>
<td>0-20 cm</td>
<td>0.1843</td>
<td>0.4581</td>
<td>0.0881</td>
<td>1.1699</td>
<td>23.49</td>
<td>0.50</td>
<td>1.4</td>
</tr>
<tr>
<td>S1</td>
<td>20-30 cm</td>
<td>0.2001</td>
<td>0.4222</td>
<td>0.0484</td>
<td>1.1656</td>
<td>46.10</td>
<td>0.50</td>
<td>1.5</td>
</tr>
<tr>
<td>S2</td>
<td>0-20 cm</td>
<td>0.1528</td>
<td>0.4896</td>
<td>0.0302</td>
<td>1.2735</td>
<td>49.22</td>
<td>0.50</td>
<td>1.4</td>
</tr>
<tr>
<td>S2</td>
<td>20-30 cm</td>
<td>0.1458</td>
<td>0.4376</td>
<td>0.0245</td>
<td>1.2620</td>
<td>75.25</td>
<td>0.50</td>
<td>1.5</td>
</tr>
</tbody>
</table>

Table 2 Soil chemical properties used in the modelling simulations. Gapon coefficients are for sand over clay soil (this report, Appendix 6).

<table>
<thead>
<tr>
<th>Soil</th>
<th>Texture</th>
<th>Exch Ca</th>
<th>Exch Mg</th>
<th>Exch Na</th>
<th>Exch K</th>
<th>CEC</th>
<th>Gapon Coefficient ($K_G$)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>---------</td>
<td>---------</td>
<td>---------</td>
<td>--------</td>
<td>-----</td>
<td>--------------------------</td>
</tr>
<tr>
<td>S1</td>
<td>Sc1</td>
<td>62.05</td>
<td>45.47</td>
<td>26.83</td>
<td>17.61</td>
<td>161.52</td>
<td>0.02</td>
</tr>
<tr>
<td>S1</td>
<td>Sc1</td>
<td>72.72</td>
<td>40.86</td>
<td>21.97</td>
<td>18.85</td>
<td>153.61</td>
<td>0.02</td>
</tr>
<tr>
<td>S2</td>
<td>Sl#</td>
<td>72.07</td>
<td>35.71</td>
<td>12.68</td>
<td>14.85</td>
<td>133.26</td>
<td>0.02</td>
</tr>
<tr>
<td>S2</td>
<td>Sl#</td>
<td>53.32</td>
<td>32.39</td>
<td>14.83</td>
<td>13.88</td>
<td>107.36</td>
<td>0.02</td>
</tr>
</tbody>
</table>

*sandy clay loam, #sandy loam
A survey of the existing greenhouses conducted in the NAP region (Awad et al., 2019) indicated four crops are commonly grown under greenhouse conditions (tomato, capsicum, cucumber and eggplant). Therefore, these crops were considered for the long-term simulations. The water quality parameters and irrigation schedules for the test crops were obtained from the IQ-QC2 model developed by Awad et al. (2019). This model generates daily crop water requirements for crops and the associated water quality parameters based on a user-defined mixing of available waters for irrigation, such as recycled water, harvested rain water and storm water. Monthly averaged irrigation water quality data for multiple years was used as shown in Table 3. Other crop specific inputs such as root water uptake parameters (Feddes, 1974) and salinity threshold-slope functions (Maas and Hoffman, 1977) are given in Appendix 1 of this report.

Table 3 Average monthly blended water quality parameters used in the UNSACHEM simulations.

<table>
<thead>
<tr>
<th>Month</th>
<th>EC</th>
<th>Ca</th>
<th>Mg</th>
<th>Na</th>
<th>K</th>
<th>Alk</th>
<th>SO4</th>
<th>Cl</th>
<th>SAR</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>dS/m</td>
<td>meq/L</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>January</td>
<td>0.30</td>
<td>0.37</td>
<td>0.53</td>
<td>2.39</td>
<td>0.02</td>
<td>0.38</td>
<td>0.73</td>
<td>2.24</td>
<td>3.6</td>
</tr>
<tr>
<td>February</td>
<td>0.42</td>
<td>0.52</td>
<td>0.74</td>
<td>3.34</td>
<td>0.02</td>
<td>0.54</td>
<td>1.02</td>
<td>3.13</td>
<td>4.2</td>
</tr>
<tr>
<td>March</td>
<td>0.41</td>
<td>0.50</td>
<td>0.72</td>
<td>3.26</td>
<td>0.02</td>
<td>0.52</td>
<td>0.99</td>
<td>3.05</td>
<td>4.2</td>
</tr>
<tr>
<td>April</td>
<td>0.37</td>
<td>0.45</td>
<td>0.65</td>
<td>2.93</td>
<td>0.02</td>
<td>0.47</td>
<td>0.89</td>
<td>2.75</td>
<td>3.9</td>
</tr>
<tr>
<td>May</td>
<td>0.30</td>
<td>0.36</td>
<td>0.52</td>
<td>2.35</td>
<td>0.02</td>
<td>0.38</td>
<td>0.72</td>
<td>2.20</td>
<td>3.5</td>
</tr>
<tr>
<td>June</td>
<td>0.22</td>
<td>0.27</td>
<td>0.39</td>
<td>1.77</td>
<td>0.01</td>
<td>0.28</td>
<td>0.54</td>
<td>1.65</td>
<td>3.1</td>
</tr>
<tr>
<td>July</td>
<td>0.16</td>
<td>0.20</td>
<td>0.28</td>
<td>1.27</td>
<td>0.01</td>
<td>0.20</td>
<td>0.39</td>
<td>1.19</td>
<td>2.6</td>
</tr>
<tr>
<td>August</td>
<td>0.17</td>
<td>0.20</td>
<td>0.29</td>
<td>1.32</td>
<td>0.01</td>
<td>0.21</td>
<td>0.40</td>
<td>1.24</td>
<td>2.7</td>
</tr>
<tr>
<td>September</td>
<td>0.23</td>
<td>0.29</td>
<td>0.41</td>
<td>1.85</td>
<td>0.01</td>
<td>0.30</td>
<td>0.57</td>
<td>1.74</td>
<td>3.1</td>
</tr>
<tr>
<td>October</td>
<td>0.21</td>
<td>0.26</td>
<td>0.38</td>
<td>1.70</td>
<td>0.01</td>
<td>0.27</td>
<td>0.52</td>
<td>1.59</td>
<td>3.0</td>
</tr>
<tr>
<td>November</td>
<td>0.15</td>
<td>0.18</td>
<td>0.26</td>
<td>1.17</td>
<td>0.01</td>
<td>0.19</td>
<td>0.36</td>
<td>1.09</td>
<td>2.5</td>
</tr>
<tr>
<td>December</td>
<td>0.11</td>
<td>0.14</td>
<td>0.20</td>
<td>0.90</td>
<td>0.01</td>
<td>0.14</td>
<td>0.28</td>
<td>0.84</td>
<td>2.2</td>
</tr>
</tbody>
</table>

The IQ-QC2 model adopted a temporally uniform 0.6 factor of open field crop evapotranspiration (ETc) to derive the crop water requirement of the glasshouse crops. The calculated ETc matches the temperature based irrigation schedule adopted by the local growers, therefore, we used these values as daily potential crop transpiration in the HYDRUS-UNSATCHEM simulations. Rainfall and soil evaporation were not considered in the simulations as these are supposed to be controlled by the presence of closed growing structures (greenhouse). The daily ETc of all the crops was estimated from the reference crop evapotranspiration (ET0) and the crop coefficient (Kc) approach (Allen et al., 1998) used for open field crops. Monthly Kc values for tomato, cucumber, capsicum and eggplant are
provided in the IQ-QC2mixing model as described by Awad et al. (2019). Climate parameters for the ETc estimation were obtained from the Bureau of Meteorology station at Edinburg Raaf (34.71°S, 138.62°E; BOM station number 023083) for the historical climate (1970-2018) and from the Goyder climate change projections for the future climate (2018-2050) (Charles and Fu, 2015).

More details on the crops, their irrigation schedule, mixing strategy, and quality of blended water are given in Awad et al. (2019). A schematic representation of the temperature based irrigation schedule for greenhouse cultivation is shown in Figure 1.

![Figure 1 Schematic representation of the temperature based irrigation schedule followed by growers in the NAP region.](image)

Initially the model was run on the crop specific temperature based irrigation schedule adopted by the growers in the NAP region (Awad et al., 2019). Later on the model was run on numerous management scenarios developed on the basis of the outcome of the initial scenario. These simulations include 4 leaching fractions (LF, in % excess water applied), i.e. 0, LF0.15 (15%), LF0.2 (20%), LF0.3 (30%) and four annual gypsum application levels, i.e. 0, 10, 15, and 20 meq/kg soil equivalent to 0, 1.7, 2.6 and 3.4 t/ha designated as G0, G10, G15 and G20, respectively. These simulations were designed to generate information on managing the available irrigation waters for their sustainable use under greenhouse cultivation in the NAP region.
3. Results and Discussion

Model simulated water balance components obtained from the base scenario based on current farmers’ irrigation practices for different greenhouse crops are shown in Figure 2. The annual values of root water uptake for all crops (tomato, cucumber, capsicum and eggplant) are close to the annual irrigation application. The average annual root water uptake by tomato, capsicum, eggplant and cucumber was 476, 642, 649, and 303 mm, respectively. However, the corresponding drainage component was only 4.6, 5.4, 6.1 and 4.1% of the total irrigation application. Such low values are below the required leaching fraction (LF) needed to flush the accumulated salts from the crop root zone. Typically, open field crop irrigation designs allow a 10-20% higher water application for salinity control depending on crop, soil and water quality. In some cases this fraction can be higher than 20% if climate, soil or crop grown encourage salinity build in the soils. Rhoades and Loveday (1990) suggested a leaching fraction of 20-50% as an ideal fraction for such conditions including for irrigation with recycled water.

![Figure 2](image)

The simulated annual values of pH, ECsw, SAR and ESP under different crops are shown in Figure 3. The pH values decreased gradually, although the magnitude of reduction was very small. At the end of the simulation at year 2050, the average pH values in the soils varied in a narrow range (8.2-8.5) for all the crops. However, pH values were relatively higher in cucumber as compared to other crops. These changes in soil pH might be related to a low pH...
of the irrigation water as compared to the soil solution, which gradually brought down the pH to achieve a quasi-equilibrium.

On the other hand, there was a sharp increase in the profile average EC\textsubscript{sw} values under all crops. However, the EC\textsubscript{sw} build up was the lowest in cucumber, followed by tomato; these results are consistent with the lower amount of seasonal irrigation applied to those two crops. Meanwhile, the highest annual EC\textsubscript{sw} was observed for capsicum and eggplant; this is the result of the higher irrigation application for these crops and a longer cropping season (Jan to Oct-Nov) as compared to cucumber (Jan- May) and tomato (Jan- Sept). The profile average EC\textsubscript{sw} at year 2050 rose to 6.5, 7.6, 8.7 and 9.3 dS/m for cucumber, tomato, capsicum and eggplant, respectively. These values are above salinity tolerance threshold EC\textsubscript{sw} (EC\textsubscript{sw} was derived from published EC\textsubscript{e} values as EC\textsubscript{sw} = 2EC\textsubscript{e}) for these crops, i.e. 5, 5, 3.4 and 2 dS/m (Maas and Hoffman, 1977; Sonneveld and Vogt, 2009).

Bonachela et al. (2018) observed that mean EC of the soil solution extracted with suction cups under a greenhouse tomato crop ranged between 6 to 7 dS/m, which is similar to the current study. However, in some cases, farmers may over irrigate to overcome the salinity problem as leaching practices are not well documented (Bonachela et al., 2018). However, soil salinization mostly depends on the amount of saline water applied (Meiri, 1984; Shalhevet, 1994). Therefore, a leaching fraction should be applied well ahead of the salt accumulation in the root environment reaching hazardous levels (Meiri, 1984; Shalhevet, 1994). Another potential factor contributing to salt accumulation is an increase in the evaporative demand due to climate variability (Fernández et al., 2010). Therefore, a soil with continued high salinity values above crop thresholds can create considerable osmotic impacts which reduces the crop water uptake and renders the soil unfit for crop production.

On the other hand, soilless production systems typically adopt a large leaching fraction (30-40%) (Bonachela et al., 2018), which is a fundamental management consideration for these growing systems to maintain the salinity of the root zone solution at levels which are not detrimental to optimal crop production (Sonneveld, 2000). To avoid problems caused by accumulation of salts, an appreciable proportion of applied nutrient solution has to be drained, the so-called “drain-to-waste” system (van Os, 1995). Therefore, soilless systems such as hydroponics represent both a substantial loss of water and potentially a major input of salts and agrochemicals to groundwater.
Similarly, annual average SAR and ESP values in the soil also showed an increasing trend for all crops. The ESP values at the end of simulation (year 2050) were 30.8, 27.1, 33.2, and 31.4 % under tomato, capsicum, eggplant and cucumber, respectively. However, the initial ESP values (13-18 %) in the soil were also high, which had increased over the model warming up period (1970- 2018) and varied for different crops in response to the amount of irrigation water applied. Undoubtedly, the final ESP values are much higher than the critical ESP (>6), which leads to the development of sodic soil conditions. Therefore, an annual addition of gypsum is essential to reclaim the soil from the high ESP values. Alternatively, a soluble Ca application as part of the irrigation above the crop requirement also helps in keeping the ESP under control. However, rapid leaching of soluble Ca or likely precipitation in the soil as calcite at high pH may reduce the effectiveness of added soluble Ca.

Similar changes in soil pH, ECsw, SAR and ESP were also observed in the sandy loam soil (S2) for all crops (Figure 4). However, the magnitude of these parameters was different for different crops. These results suggest that the temperature based irrigation schedule for
greenhouse crops in the NAP region appears not to suffice to control the buildup of salinity and sodicity hazards in the soil, especially if such schedule is followed for a long time without any management options. Additionally, soil salinity normally arises if moderately brackish or saline waters are used for irrigation, as frequently occurs when recycled water or groundwater are used for irrigation or blending. Therefore, there is need to optimize the irrigation schedule with an appropriate leaching fraction (LF) as well as by applying suitable amounts of gypsum (G) for controlling salinity and sodicity.

Additional scenarios were performed with leaching fractions (LF0.15, LF0.2, and LF0.3) and annual gypsum applications (G10, G15, G20) for soil grown greenhouse tomato crop. Results show that an increased leaching fraction alone (i.e. without gypsum) has a small impact on soil pH, with values being reduced by 0.3-2.4% only when the LF increases from 0.15 to 0.3, respectively (Figure 5). However, gypsum addition at an annual rate of 10 meq/kg soil (G10, 1.7 t/ha) reduced the soil pH at all depths by 7.5-8 %. However, further increase in the gypsum application (G15, G20) had little impact on subsequent reductions in the soil pH. Thus, the average pH has been reduced to 7.74-7.8, which may not have any adverse impact on the
nutrient availability for crops. Therefore, an annual gypsum application of 10 meq/kg soil (1.7 t/ha) seems adequate, at least to maintain the soil pH within a suitable range.

Figure 5 Model simulated annual profile average a) pH, b) ECsw, c) SAR, and d) ESP in the sandy clay loam soil (S1) under different leaching fractions (LF0, LF0.15, LF0.2, and LF0.3) and annual gypsum application (G0, G10, G15 and G20) scenarios for tomato cultivation under greenhouse condition.

Similarly, the ECsw in the soil decreased below the tomato threshold (ECsw = 5 dS/m) for all LFs (0.15 to 0.3), which suggests that a minimum LF of 0.15 is adequate to maintaining the soil salinity below the threshold (Figure 5). Interestingly, application of gypsum increased the soluble ion concentration in the soil, leading to an increase in the ECsw at all depths, although it was still below the crop threshold. However, it is worth noting that increasing the LF alone didn’t reduce the SAR and ESP in the soil. Apparently, these impairments require addition of soil amendments such as gypsum to reduce their impact on soil as well as on crops. Therefore, an addition of gypsum at an annual rate of 10 meq/kg soil (1.7 t/ha) along with the leaching fraction LF 0.15 reduced the SAR and ESP below threshold within the 30 cm soil depth. However, this combination was unable to bring down SAR and ESP below threshold levels at 60-90 cm depth. Further increasing the gypsum levels to 15 (G15) and 20 meq/kg (G20) combined with the leaching fraction LF 0.15 also did not further reduce SAR and ESP in the soil at deeper depths. This suggests that larger leaching fractions are required to remove excess salt and obtained acceptable SAR and ESP values at lower soil depths.

A further increase in the LF to 0.2 combined with and annual addition of gypsum at 10 meq/kg soil (1.7 t/ha) has reduced the average SAR and ESP below thresholds, also at lower soil depths.
Other scenarios were also explored in which leaching fractions less than 0.15 were used together with different gypsum levels (G10, G15 and G20): while the ECsw dropped below threshold, they were unable to bring down the ESP below the critical threshold (data not shown). The conclusion is that, for shallow rooted soil grown vegetables, addition of gypsum at a rate of 10 t/ha with 0.15 LF is able to keep the soil free from salinity and sodicity hazards under greenhouses conditions. On the other hand, for deep rooted crops, a combination of 0.2 LF with a 10 meq/kg soil or higher annual gypsum addition would help in maintaining sustainable production under long-term greenhouse conditions.

An annual comparison of profile average pH, ECsw, SAR and ESP values obtained with and without 0.2 LF along with different annual gypsum additions (0, 10, 15 meq/kg soil) for tomato cultivation is shown in Figure 6. The G10 with 0.2 LF scenario initially had average profile SAR and ESP values higher than the threshold (< 6). These reduced gradually to the values similar to those obtained with 15 meq/kg soil (G15) gypsum at the end of the simulation at year 2050. Similarly, the average SAR values were also reduced below the threshold, the pH reduced by 8.7% with average values around 7.7, and ECsw was much lower than the tomato tolerance threshold. Similar results were obtained in the sandy loam soil (S2) with cucumber (Figure 7). However, the magnitude of variation between G10LF0.2 and G15LF0.2 scenarios was much lower as compared to tomatoes. Apparently, the amount of irrigation application and soil properties also influences the effectiveness of LF and gypsum addition for reclaiming the soils. Other factors such as purity of gypsum, particle size, uniformity of application and mixing in the soil also influence the effectiveness of gypsum in reducing the ESP in the soil. These factors were not evaluated in this study.
Overall, the results from the current modelling exercise suggest that a long-term irrigation schedule without leaching fraction may lead to salinity and sodicity problems in the soil. There
is clearly a need to apply a leaching fraction of 15-20% for leaching the salts below the crop root zone. Even the blended water used for irrigation has a strong tendency to build up sodicity hazards in the soil. Therefore, annual applications of gypsum at 1.7-2.6 t/ha (or preferably on the basis of soil a test) or other sources such as a Ca solution should be applied to overcome the sodium hazard in the soil. The survey conducted by Awad et al. (2019) found that many growers apply organics/compost annually and also apply Ca through nutrient solutions. Both of these additions help in alleviating sodicity hazards in the soil. However, continued monitoring of the salinity and sodicity status of the soil is an essential aspect of soil management and allows one to intervene well before issues occur.

The results obtained from the current study are specific to the crops, soils, climate, and water quality considered. Several other aspects of protected crops were not considered in this study, such as different substrates, effect of ventilation and evaporative cooling. These can also impact the extent of water use and its management, and related issues such as salinity and sodicity development. Concentration of nutrient solution, quality of substrate and local issues such as leachate management may also impact the plant water requirement and its effective management. Simulation tools such as those used in this study can account for these additional conditions and therefore help develop guidelines for practitioners aimed at maintaining sustainable production systems.

4. Conclusions

Protected agriculture is a popular cultivation system especially for vegetable crops. They are highly efficient systems in term of energy and other input consumption such as water and fertilizers. However, suitable water is often scarce in the regions where these production systems exist, such as in the vicinity of urban, arid or semi-arid environments. Therefore, in most cases, low quality water is used directly or in combination with other available water resources such as harvested rainwater. Unfortunately, low quality waters are often used without knowing the harmful impacts on crop productivity, soil health, and the wider environment.

This study used the multi-component major ion chemistry module UNSATCHEM of the HYDRUS-1D model to evaluate the effects of long-term (2018-2050) irrigation with blended water for soil grown tomato, cucumber, capsicum and eggplant under unheated greenhouse conditions. The results revealed that irrigation schedules that do not apply a significant leaching
fraction may lead to high salt build up and ESP development in the soil while accounting for future climate projections. The soil solution salinity (ECsw) can increase to 6.5-9 dS/m at year 2050 and ESP can rise to 27-33% for all crops considered. These conditions could render the soil unfit for crop production and could potentially degrade the associated environment. Therefore, appropriate management options should be implemented to keep the irrigation induced harmful impacts under control.

The study evaluated the efficacy of increased leaching and gypsum addition to control salinity and sodicity. For this purpose the UNSATCHEM module was extended with a new capability to allow annual gypsum applications. Management scenarios with different leaching fractions for salinity control showed that 15-20% more water per irrigation would be required to keep the salinity under control for soil grown greenhouse vegetables. Results obtained in various scenarios for amelioration of soil with high ESP suggest that annual gypsum application at a rate of 1.7 t/ha was adequate for managing this hazard. However, both management options (i.e. leaching fraction and gypsum use) need to be implemented simultaneously. Finally, long-term monitoring of highly efficient greenhouse production systems is essential for early identification of irrigation induced soil issues.

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Appendix 10 Optimising the riparian zone width near a river for controlling lateral migration of irrigation water and solutes

Contents

<table>
<thead>
<tr>
<th>Section</th>
<th>Page</th>
</tr>
</thead>
<tbody>
<tr>
<td>Executive Summary</td>
<td>349</td>
</tr>
<tr>
<td>1. Introduction</td>
<td>350</td>
</tr>
<tr>
<td>2. Materials and Methods</td>
<td>352</td>
</tr>
<tr>
<td>2.1. Experimental site</td>
<td>352</td>
</tr>
<tr>
<td>2.2. Soil characteristics</td>
<td>353</td>
</tr>
<tr>
<td>2.3. Water resources</td>
<td>354</td>
</tr>
<tr>
<td>2.4. Vegetation and crops parameters</td>
<td>355</td>
</tr>
<tr>
<td>2.5. Irrigation scheduling of crops</td>
<td>356</td>
</tr>
<tr>
<td>2.6. Transport domain, initial and boundary conditions</td>
<td>357</td>
</tr>
<tr>
<td>2.7. Calibration and validation of the model</td>
<td>358</td>
</tr>
<tr>
<td>2.8. Scenario analysis</td>
<td>359</td>
</tr>
<tr>
<td>2.9. Model evaluation</td>
<td>359</td>
</tr>
<tr>
<td>3. Results and Discussion</td>
<td>360</td>
</tr>
<tr>
<td>3.1. Calibration and validation of HYDRUS-2D</td>
<td>360</td>
</tr>
<tr>
<td>3.2. Irrigation and annual water balance for crops</td>
<td>362</td>
</tr>
<tr>
<td>3.3. Hydraulic exchange at the river-buffer interface</td>
<td>362</td>
</tr>
<tr>
<td>3.4. Impact of a buffer width on solute dynamics</td>
<td>365</td>
</tr>
<tr>
<td>4. Implications of the findings</td>
<td>370</td>
</tr>
<tr>
<td>5. Conclusions</td>
<td>370</td>
</tr>
<tr>
<td>References</td>
<td>371</td>
</tr>
</tbody>
</table>
Executive Summary

Riparian zones are essential to preserve water quality of rivers adjacent to large areas of irrigated agriculture. We used HYDRUS (2D/3D) to quantify the long-term (1st July, 2009 to 30th June, 2017) influence of crops (almonds, wine grapes and potato-carrot) irrigated with DAFF/recycled water (RCW) on water and solute exchange at the Gawler River interface in relation to vegetation buffer widths from 10-110 m. The model was calibrated and validated for groundwater table dynamics near the river considering all hydrologic fluxes occurring in the domain such as rainfall, irrigation, evapotranspiration by crops and the vegetation buffer, as well as groundwater fluctuations and daily water level changes in the river. The major findings includes:

- The hydraulic exchange at river interface for different irrigated crops was found to be sensitive to the buffer widths.
- The likely average annual water flow from the almond and annual horticulture irrigated area to the river was nearly twice as much (2.1 and 1.8, respectively) that under wine grapes.
- For wine grapes, almonds and annual horticulture, the average annual hydraulic balance reached an equilibrium at 20, 65 and 55 m buffer widths, respectively.
- The average annual load of salts became negligible for wine grapes with a 20 m buffer width.
- This study shows that buffer widths of 20, 60, and 40 m for irrigated wine grapes, almond, and annual horticulture, respectively, are needed to restrict the migration of salts to the river.

The optimised widths in this study differs from the existing guidelines in Australia. It is suggested that there is a strong need to revise the existing riparian width guidelines for maintaining good water quality in surface water bodies near RCW irrigated crops. Further refinements are possible by incorporating the influence of preferential flow paths, improved water stress response functions, and addressing the data limitations for calibration of the model for solute dynamics.
1. Introduction

Crop production, especially in arid and semi-arid regions of the world where rainfall is not able to meet the evapotranspiration needs of the crops, depends on supplemental irrigation. Irrigated agriculture contributes 40% of the world food production from 20% of the cropped area, thus makes a major contribution to the global food security (FAO, 2016). However, the use of low water quality (e.g. recycled water, RCW) for irrigation can induce numerous adverse impacts. Irrigated agriculture with RCW water may become unsustainable due to its contribution to soil degradation, salinization, waterlogging, and environmental pollution. Global water security warrants beneficial reuse of recycled water, such as irrigation, but with minimal potential harmful impacts on ecosystems. Ecosystem impairment, particularly reduced soil quality, biodiversity loss, and harm to amenity and cultural heritage values, is a growing global problem (FAO, 2011). Therefore, future irrigation schemes must address trade-offs, particularly with respect to inter-sectoral water allocations and environmental impacts.

Aquatic ecosystems adjacent to irrigated agriculture are most at risk due to the transport of irrigation induced chemicals such as soluble salts, nitrates, and pesticides (Zhang et al., 2010). The fate of these chemicals in the soils and their migration to receiving environments depend on a number of factors including the vegetation, topography, climate, soil, irrigation, groundwater level, and flow conditions in the stream (Klatt et al., 2017; Schilling et al., 2018). Riparian vegetation can moderate the movement of water and solutes to water bodies by interception and attenuation of chemicals moving through the buffer zone (King et al., 2016).

The Australian Water Act 1989 designated the riparian zone as 20 m on either side of a waterway and does so for the 'declared' delivery of stability, conservation, or functioning of the waterway. However, the Assessment of River Condition (ARC) found that more than 80% of Australian rivers and riparian lengths are affected by human-generated catchment disturbance (Norris, 2001). Rassam et al. (2006) reported that the vegetative buffers along many perennial and ephemeral streams varied from 15 to 20 m. It is of some concern that the Water Act 2007 no longer makes specific references to riparian zones. Therefore, some Australian states devised their own buffer zone guidelines, depending on the priority areas and water quality protection needs. For example, in Western Australia (Department of Water, Government of Western Australia, 2006), the minimum buffer width guidelines for protection of margins of water supply sources, drains, and banks of the first and second order ephemeral streams and wetlands is 30 m. On the other hand, the New South Wales (NSW) guidelines
Appendix 10 Optimising the riparian zone width along a river | 351

(Department of Primary Industries, NSW, 2012) suggest the buffer widths (one side of a watercourse) depend on the type of watercourse, i.e., 1st order (10 m + half channel width), 2nd order (20 m + half channel width), 3rd order (30 m + half channel width), and 4th order including estuaries and wetlands (40 m + half channel width). Similarly, the guidelines in Tasmania suggest a 30 m width of native vegetation as a riparian boundary for wetlands and waterways. Riparian widths for Queensland irrigators for maintaining good water quality is 30 m (Department of Environment and Resource management, 2011). In Victoria, the guidelines depend on the land use intensity, and for the high intensity land use such as irrigated crops, the buffer width is 60 m for maintaining water quality in surface water bodies. Essentially, the Victorian guidelines are more comprehensive and clearly define the management objective for riparian widths (Hansen et al., 2010). To our knowledge, South Australia does not have any guidelines for maintaining an appropriate riparian width for preventing subsurface migration of salts and contaminants from irrigated agriculture to surface water bodies’ especially seasonal rivers.

Several investigations have examined the functions of buffer zones for stream ecosystems (e.g., Mayer et al., 2007; Hansen et al., 2010). However, these have mostly dealt with the overland movement of solutes via surface runoff and sediment transport. Only, a limited number of modelling or case studies have evaluated the role played by buffer zones in reducing the migration of irrigation induced soluble salts/contaminants via subsurface flow to streams (e.g., Naiman et al., 2005; Allaire et al., 2015). Subsurface flow paths can exhibit wide variations depending on specific local conditions (Naiman et al., 2005) including subsurface lithology and stratigraphy (DeVito et al., 2000; Hill et al., 2004). To our knowledge, no information is currently available in the literature on the role played by buffer zones in dealing with the irrigation induced solute interception or influencing its migration to water bodies.

Field experiments for assessing the role of a buffer zone on the subsurface water and salts movement from irrigated cropping system to an adjoining river is both a complex and expensive exercise. Therefore, numerical models are increasingly being used (e.g., Flipo et al., 2014; Xian et al., 2017) for such assessment. Hydraulic exchange across the stream-aquifer has been modelled with buffer zones (e.g., Phogat et al., 2017a) or without (e.g., Baratelli et al., 2016). Similarly, Kidmose et al. (2010) employed a conceptual groundwater flow and reactive transfer model to establish a relationship between flow paths and the fate of a pesticide in a riparian wetland. Alaghmand et al. (2013) used a numerical model (Hydrogeosphere) to
evaluate the interaction between a river and a saline floodplain in relation to groundwater fluctuations, incorporating evapotranspiration losses by riparian vegetation. Klatt et al. (2017) explored the capability of a coupled hydro-biogeochemical model to evaluate the effectiveness of buffer strips to reduce nitrogen loads into aquatic systems. However, most of these modelling studies have been either conceptual and/or only partially calibrated for site-specific flow and/or solute dynamics. The complicated nature of water and solute transport processes (Sophocleous, 2010) and the inherent uncertainty of input data are some of the challenges in simulating water flow and solute transport with physically based models. Nevertheless, such models are valuable in understanding water flow and solute transport/reaction processes involved in complex bio-geological environments.

This study uses a two-dimensional finite element numerical model HYDRUS (2D/3D (referred to below as HYDRUS; Šimůnek et al., 2016) to quantify the extent of water and solute exchange across a stream-buffer interface. The study involves complex heterogeneous geological formations involving real-time climatic, vegetative (crop and buffer), and stream flow conditions. The key objectives of this investigation were: i) to calibrate and validate a numerical model (HYDRUS) for water table dynamics in an area adjacent to a seasonal river (Gawler River) by incorporating daily water level fluctuations in the river, groundwater dynamics, crop evapotranspiration, riparian zone vegetation evapotranspiration, and soil heterogeneities; ii) to estimate the impact of different buffer zone widths on the flux exchange at the river-buffer interface under different cropping systems, iii) to optimise the riparian width to control the irrigation-induced solute movement to the river for different irrigated crops; and iv) to estimate the residence time of the solute tracer migrating to the adjoining water body through the subsurface under shallow water table conditions.

2. Materials and Methods

2.1. Experimental site

The study was carried out at the Virginia Park (34°38ʹ22.6ʺS and 138°32ʹ27.6ʺE) gauging station at Gawler River which is situated at 12 m above the Australian Height Datum (AHD). The Gawler River only flows during the rainy season (July to October). However, stagnant water (about 30-100 cm)/base flow conditions prevail at other times at the gauging station. The adjacent area, being a part of the vast Northern Adelaide Plains (NAP), has a relatively flat
topography with a gentle slope to the west. All relevant features of the study site are shown in Figure 1. The NAP experiences a Mediterranean climate, which is characterised by hot, dry summers and cool to cold winters. Long-term (1900-2016) average rainfall in the region amounts to 475 mm (Department of Environment, Water and Natural Resources, 2016) and annual evapotranspiration amounts to 1308 mm, resulting in the irrigation demand for crop production. Water table fluctuations in the area adjacent to the river were monitored in the shallow wells. Location of these wells is shown in Figure 1.

Figure 1. Map of the study site showing the Gawler River, the gauging station, shallow wells (yellow circles) and adjacent cropped area.

2.2. Soil characteristics

The soils of the NAP are highly heterogeneous with depth. There is commonly a shallow clay layer at a variable depth, which determines the root growth and crops to be grown. Broader soil groups and geology of the site were obtained from the stratigraphic information of the site and well logs within the vicinity of the site. There are in general 6 major geological layers, which include red friable sandy loam soil, light brown silty topsoil, sandy clay, sandy non-calcareous clay, non-calcareous fine sandy clay, and sand. The soil particle size distributions and bulk densities of these soil groups were obtained from the previous soil analysis reported in ASRIS (ASRIS, 2011) and the APSIM (Holzworth et al., 2014) data base. The particle size and bulk density data were used to estimate soil hydraulic parameters using the ROSETTA module embedded in the HYDRUS software environment. The saturated hydraulic conductivity ($K_s$), and the $\alpha$ and $n$ parameters were further adjusted during the calibration process and their final optimised values are presented in Table 1.
Table 1. Optimised soil hydraulic parameters used in numerical simulations.

<table>
<thead>
<tr>
<th>Sr No</th>
<th>Textural class</th>
<th>Depth (m)</th>
<th>$\theta_r$ (cm$^3$ cm$^{-3}$)</th>
<th>$\theta_s$ (cm$^3$ cm$^{-3}$)</th>
<th>$\alpha$ (cm$^{-1}$)</th>
<th>n</th>
<th>$K_s$ (cm d$^{-1}$)</th>
<th>l</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Sandy loam</td>
<td>0-1</td>
<td>0.07</td>
<td>0.44</td>
<td>0.024</td>
<td>1.45</td>
<td>56.1</td>
<td>0.5</td>
</tr>
<tr>
<td>2</td>
<td>Silty loam</td>
<td>1-2</td>
<td>0.06</td>
<td>0.45</td>
<td>0.03</td>
<td>1.5</td>
<td>69.8</td>
<td>0.5</td>
</tr>
<tr>
<td>3</td>
<td>Clay loam</td>
<td>2-5.5</td>
<td>0.07</td>
<td>0.43</td>
<td>0.0227</td>
<td>1.41</td>
<td>39.9</td>
<td>0.5</td>
</tr>
<tr>
<td>4</td>
<td>Silty clay loam</td>
<td>3.5-6</td>
<td>0.08</td>
<td>0.44</td>
<td>0.023</td>
<td>1.30</td>
<td>25.0</td>
<td>0.5</td>
</tr>
<tr>
<td>5</td>
<td>Sandy clay</td>
<td>6-8</td>
<td>0.09</td>
<td>0.44</td>
<td>0.0234</td>
<td>1.26</td>
<td>18.8</td>
<td>0.5</td>
</tr>
<tr>
<td>6</td>
<td>Sand</td>
<td>8-12</td>
<td>0.05</td>
<td>0.41</td>
<td>0.124</td>
<td>2.28</td>
<td>350.0</td>
<td>0.5</td>
</tr>
</tbody>
</table>

2.3. Water resources

There are three main water sources used for irrigation in the NAP: surface water (the Gawler River), groundwater from tertiary aquifers, and recycled water from the Bolivar sewage treatment plant. Water resources in the NAP Prescribed Wells Area (PWA) are managed through a water allocation plan. The largest reserves of fresh groundwater in the NAP are contained in the Tertiary aquifers, i.e., T1 (the average thickness 60-70 m) and T2 aquifers (the average thickness 80-120 m), below the 60 m depth (Gerges, 1999). Groundwater extractions primarily occur from the T2 aquifer, which provides 73% of the total groundwater volume, while the T1 aquifer is less developed (25%). The Quaternary aquifers provide the remaining 2% (Department of Environment, Water and Natural Resources, 2017a, b). The shallow Quaternary (Q1 to Q4) aquifers have only a limited water source and usually high salinity (2000 mg/L to 15,000 mg/L), which restrict them from being used for agricultural consumption. The groundwater flow direction in all Quaternary aquifers is generally from the Mount Lofty Ranges (MLR) in the east toward the Gulf of St. Vincent in the west (Lamontagne et al., 2015). Groundwater recharge to the tertiary aquifers is thought to occur by lateral inflow from the adjacent fractured rock aquifers of the MLR as direct recharge from rainfall in the NAP is low (Bresciani et al., 2017). The groundwater level observation network for the NAP PWA consists of 153 observation wells monitored at quarterly intervals (Department for Water, South Australia, 2010). Sixty-one wells are within Quaternary aquifers and 82 are within Tertiary aquifers. The groundwater salinity observation network for the NAP PWA consists of 40 observation wells monitored on an approximately annual basis (Department for Water, South Australia, 2010).
The Gawler River downstream of the junction of the North and South Para Rivers, is a Prescribed Watercourse in the Western MLR Prescribed Water Resources Area. Extraction limits and minimum and maximum threshold flows rates have been set through the Western MLR WAP (Department of Environment, Water and Natural Resources, 2016). Specifically, 10 GL per annum is the extraction limit defined in the WAP, with water allowed to be extracted at flows rates of between 500 to 690 L/s.

Water resources for irrigated agriculture in the NAP have been supplemented by 19.5GL from the Bolivar tertiary-treated wastewater plant since 1999. This recycled water supplies 400 irrigators (The Goyder Institute of Water Research, 2016). An additional 12 GL per year of recycled water will soon be available to increase the area under irrigation. The recycled water is a Class-A treated sewage water but contains on an average 1200 mg/L soluble solutes (Stevens et al., 2003). Managed aquifer recharge (MAR) schemes have been operational in the NAP for some decades, with a number of new wells established (e.g., Bolivar). The total current injection rate is estimated to be 8.5 GL yr⁻¹ (Bresciani et al., 2017).

2.4. Vegetation and crops parameters

The vegetation buffer at the study site is dominated by river red gums (Eucalyptus spp.) but its width is highly variable along the longitudinal distance of the river, ranging from a couple to hundreds of metres. The area adjoining the riparian buffer is used for intensive cropping such as almonds, wine grapes, potato, carrot, and onion all along the river. On the southern side of the river where the modelling domain was established, the land has been used for the wine grape cultivation.

Daily crop evapotranspiration \((\text{ET}_c)\) for river red gum \((\text{Eucalyptus spp.})\) in the buffer zone and irrigated crops (wine grape, almond, carrot, potato) grown in the adjacent river corridor were estimated from daily reference crop evapotranspiration \((\text{ET}_0)\) data and local crop coefficients \((K_c)\). The \(\text{ET}_0\) (Allen et al., 1998) data was obtained from the nearby weather station (Edinburg Raaf). The daily \(\text{ET}_c\) values were divided into evaporation \((\text{E}_s)\) and transpiration \((\text{T}_p)\) components based on the leaf area index (LAI) as follows (Ritchie, 1972):

\[
\text{E}_s = \text{ET}_c \cdot e^{K_p \times \text{LAI}}
\]

\[
\text{T}_p = \text{ET}_c - \text{E}_s
\]
Here, $K_{sr}$ is the light extinction coefficient for global solar radiation and its value was set to 0.5 (Aubin et al., 2000; Phogat et al., 2017a) for all vegetations. The LAI data for wine grapes and almonds was obtained from other studies (Phogat et al., 2017b, 2018b) and for the annual horticulture (carrot and potato) crops from the literature (Reid and English, 2000; Deshi et al., 2015). Canopy interception by the buffer vegetation was assumed to be 15% of precipitation (Xiao et al., 2000). Estimated daily $E_s$ and $T_p$ values and daily rainfall were used as input in HYDRUS simulations.

The roots of the buffer strip vegetation were assumed to be distributed linearly from the soil surface to a depth of 200 cm. Although roots of *Eucalyptus* can grow to a depth of 6-7 m (Phogat et al., 2017a), however, due to shallow water table conditions at the site, roots generally did not grow far below a water table due to the lack of the oxygen supply (Baker et al., 2001). Similarly, the rooting depths of 100, 200, and 60 cm for wine grape, almond, and annual horticulture (carrot and potato), respectively, were used in the modelling study based on relevant studies from the region (Phogat et al., 2017b, 2018b). The root water uptake parameters for almond and wine grape were also taken from these studies and the HYDRUS database (potato and carrot). Since HYDRUS does not allow using different parameters for stress response functions for the crop and buffer zones, hence the same Feddes’ parameters (Feddes et al., 1978) were used for both parts of the domain. It must be noted that root water uptake in HYDRUS depends on the availability of water in the soil, the root spatial distribution, and differential transpiration fluxes in the crop and buffer zones. The root water uptake was assumed to be linearly distributed with depth, with the maximum at the soil surface and zero at the bottom of the rooting zone.

### 2.5. Irrigation scheduling of crops

The “trigger irrigation” option available in HYDRUS (Dabach et al., 2013) was used to generate irrigation schedules for all crops (wine grape, almond, annual horticulture). Irrigation is triggered when the predefined suction level in the soil profile is reached at a specified location. The timing of such an event depends on the daily climate conditions, plant water requirements, soil texture, and water availability in the soil profile. The trigger pressure used for wine grapes, almond, and carrot-potato were -60, -25, and -15 kPa, respectively. Similar trigger pressures have been used for these crops in other studies (Green, 2010; Phogat et al., 2018a, b). Similar irrigation scheduling using tensiometers is widely used for different crops.
The trigger point in this study was located at a depth of 30 cm in the middle of the irrigated crops area. Triggered irrigations had a solute concentration of 1200 mg/L (Stevens et al., 2003), which is equal to the average quantity of total dissolved solids in the Class-A treated recycled water from the Bolivar treatment plant, which is used for irrigation in the NAP region.

2.6. Transport domain, initial and boundary conditions

The transport domain represents a 400 m cross section from the middle of the river (Figure. 2). The vertical dimension represents the distance from the Australian Height Datum (AHD) to the soil surface (12 m) at the experimental site. The top width of the river was 10 m, the bottom width 4 m, and the depth 4 m at the study site. The width of the buffer zone is 30 m from the river bank. Therefore, the lateral width of the riparian zone at the Virginia Park gauging station from the middle of the river is approximately 35 m, which also includes an unsealed road which runs along the river. The finite element discretization resulted in 10,000 2D elements in a standard rectangular 2D domain.

On the upper left side of the domain (Figure. 2), the atmospheric boundary was considered through which the infiltrative influx or the evapotranspirative efflux occurs. A time-variable flux boundary condition (treated similarly as an atmospheric boundary condition) was imposed on the upper right side of the domain to represent the buffer zone, which had different fluxes than the irrigated surface. The flux at this boundary was given by the difference between daily rainfall and daily potential evaporation ($E_s$). A special HYDRUS boundary condition (BC) was specified in the river. This special BC assigns the hydrostatic pressure head on the boundary below the water level in the river and a seepage face BC on the boundary above the water level. The specified water levels in the river are linearly interpolated in time in order to smooth the impact of daily fluctuations of water levels in the river (Phogat et al., 2017a). Measured values of water table depths in the well near the left boundary of the domain (PTA100) were used to define initial and time-variable pressure head boundary conditions. No flow was assumed as the boundary. The initial pressure head condition in the domain was specified by interpolating measured mean water table depths in the shallow wells (Figure. 1) in the adjacent area while considering hydrostatic equilibrium conditions in the vertical direction. Daily rainfall in excess of the soil infiltration capacity is accounted for as run off by HYDRUS. The longitudinal dispersivity was assumed as one tenth of the modeling domain (with the transverse dispersivity
being one tenth of the longitudinal dispersivity) (Cote et al. 2003; Phogat et al., 2014) and the molecular diffusion coefficient in water equal to 1.66 cm$^2$/day (Phogat et al., 2018b).

**Figure 2.** Schematic representation of the flow domain showing the material distribution, the river, the buffer strip, irrigated crops, and imposed boundary conditions.

### 2.7. Calibration and validation of the model

Measured water table depths (average of four quarterly measurements in a year) in the shallow wells (PTA101, PTG080 and PTG087, Figure 1) near the study site were used for the calibration and validation of the model. Simulations were carried out for 1461 days (1$^{st}$ July 2009 to 30$^{th}$ June 2013) to calibrate the model for water table depths at the middle of the domain ($X = 200$ m). For most sensitive model parameters including the saturated hydraulic conductivity ($K_s$), and the coefficients $\alpha$ and $n$ of different soil layers were varied manually and no automated parameter optimisation procedure was used to calibrate the model. In addition to a visual comparison of observed and simulated water table depths, a quantitative evaluation of the model performance was undertaken using goodness-of-fit measures (see section 2.8) similar to other studies (e.g., Alaghmand et al., 2013, 2014). The calibrated model was validated for 1461 days (1$^{st}$ July 2013 to 30$^{th}$ June 2017) by comparing the measured and simulated water table depths. The calibrated and validated model was then used to assess the impact of other irrigated crops (almond and annual horticulture crops such as carrot and potato) and the buffer zone widths on the migration of water and solutes to the river. More details on different scenarios are given below in the Scenario Analysis section.

To understand the movement of irrigation-induced solutes/ agrochemical tracers to the river water, we considered Total Dissolved Solids (TDS) as representative of all soluble solutes which is consistent with numerous studies (e.g., Ramos et al., 2011; Phogat et al., 2014, 2018b). The initial soil conditions in the domain were assumed to be solute free. The average quantity
of TDS (1200 mg/L) of Class-A treated (recycled) water from the Bolivar treatment plant (Stevens et al., 2003) was applied during all triggered irrigations at the atmospheric boundary where crop is being grown. However, the model calibration for solute dynamics could not be conducted due to the non-availability of site-specific data for solute transport processes.

2.8. Scenario analysis

The calibrated and validated model was then used to simulate the dynamics of the hydrological fluxes and solute movement for different buffer widths and for various irrigated crops (wine grape, almond, and carrot-potato rotation). The simulations were executed for 8 years (1st July 2009 to 30th June 2017) plus further 8 years if needed (if solute did not reach the river) for all 3 irrigated crops for varying buffer zone widths (10 - 110 m) from the centre of the river. These simulations were established to evaluate the appropriate width of the riparian zone to control the lateral movement of solutes to the river. In the simulations extended in future climate (8 years), median climate change data for the Edinburg RAAF station were used (Charles et al., 2015). Initial conditions and daily water level fluctuations for such future simulations were imported from the previous simulations (8 years). To estimate the extent of leaching, the annual water balance for different crops was computed using inputs such as rainfall, irrigation, and model-simulated evaporation and transpiration and assuming similar initial pumping well and river exchange conditions in the domain.

2.9. Model evaluation

The model’s performance was evaluated by comparing measured \( (M) \) and HYDRUS simulated \( (S) \) water table depths. Correlation coefficients were estimated to understand the relationship between measured and simulated values of water table depths during the calibration and validation periods. The statistical error estimates [mean error \( (ME) \), mean absolute error \( (MAE) \), and root mean square error \( (RMSE) \)] between the measured and simulated water table depths were estimated as:

\[
ME = \frac{1}{N} \sum_{i=1}^{N} (M_i - S_i) \tag{2}
\]

\[
MAE = \frac{1}{N} \sum_{i=1}^{N} |M_i - S_i| \tag{3}
\]
\[ RMSE = \sqrt{\frac{1}{N} \sum_{i=1}^{N} (M_i - S_i)^2} \]  

(4)

Many studies (e.g., Coffey et al., 2004; Alaghmand et al., 2013) have used similar goodness of fit measures (correlation coefficients and RMSE) as above.

We also evaluated the test of significance between the measured and simulated values of water table depths using the paired \( t \)-test (\( t_{cal} \)) as below:

\[ t_{cal} = \frac{d}{SD_m \sqrt{\frac{1}{n}}} \quad d = \bar{M} - \bar{S} \quad \text{and} \quad SD_m = \sqrt{\frac{ns_1^2 + ns_2^2}{2n-2}} \]  

(5)

Here, \( n \) is the number of comparable paired points, \( s_1 \) and \( s_2 \) are the standard deviations of measured and simulated data, respectively, \( d \) is the difference between measured (\( \bar{M} \)) and simulated (\( \bar{S} \)) means values, \( SD_m \) is the standard deviation of the mean, and \( t_{cal} \) is the calculated paired \( t \)-test value. The null hypothesis tests that there is no significant difference between the mean values of measured and simulated water table depths.

Nash and Sutcliffe (1970) model efficiency (\( E \)) is a normalized statistic that expresses the relative magnitude of the residual variance compared to the variance of the measured data during the period under investigation, as given below:

\[ E = 1 - \frac{\sum_{i=1}^{N} (M_i - S_i)^2}{\sum_{i=1}^{N} (M_i - \bar{M})^2} \]  

(6)

The range of \( E \) lies between \( -\infty \) and 1.0 (a perfect fit). An efficiency value between 0 and 1 is generally viewed as an acceptable level of performance. Efficiency <0 indicates that the mean value of the observed time series would be a better predictor than the model and denotes unacceptable performance (Moriasi et al., 2007; Legates and McCabe, 1999).

3. Results and Discussion

3.1. Calibration and validation of HYDRUS-2D

The measured buffer zone width (35 m from the middle of the Gawler River) and actual crop grown (wine grape) adjoining the study site were considered for the calibration and validation simulations executed from 1st July, 2009 to 30th June, 2013 and 1st July, 2013 to 30th June, 2017, respectively. The data in Figure. 3 demonstrates a consistent performance of the model (i.e. \( R^2 \) of 0.66 and 0.64, and \( E = 0.34 \) and 0.34, respectively) during calibration (2009-2013)
and validation (2013-2017) period. These values fell within the $R^2$ values (0.35-0.84) reported in other modelling studies (e.g., Coffey et al., 2004; Phogat et al., 2016). Other statistical estimates ($ME$, $MAE$, $RMSE$ and $SD$; see Figure 3) during the calibration and validation period were similar but slightly higher than previously observed values (e.g. Alaghmand et al., 2013). This is because of wide fluctuation within the input data. The paired $t$-test indicated that there was no significant difference ($p=0.05$) between the measured and simulated mean depths of water table. Overall, all these statistics confirm an adequate representation of groundwater fluctuations by the model.

![Figure 3](image-url)

**Figure 3.** Relationship between measured and simulated water table depths, statistical error estimates ($ME$, $MAE$ and $RMSE$), standard deviation ($SD$) and model efficiency ($E$) values during the calibration (2009-13) and validation (2013-17) periods.

Calibration of the model for solute dynamics was not possible due to the lack of site specific data. However, numerous studies have shown that sufficiently calibrated and validated model for complex hydraulic fluxes in a heterogeneous domain can offer valuable practical understanding of bio-geochemical processes in the soil. For example, Alaghmand et al. (2013, 2014) indicated that the complexity associated with the quantification of the solute transport parameters restricted them from validating the model on the observed concentration pattern. However, their study unravelled the impacts of groundwater pumping on salinization risks of a flood plain. Similarly, Carr et al. (2018) calibrated and validated a 2D model on gauged groundwater elevations and hypothesized that the accurate representation of flow dynamics can inform environmental management involving transport of sediments, nutrients, and heavy metals. Other studies also have used similar approaches (e.g., Rousseau et al., 2012).
3.2. Irrigation and annual water balance for crops

The extent of average annual irrigation among all crops during 2009-2017 (Figure 4) was the lowest in wine grape (242 and 320 mm), followed by almond (760 and 920 mm) and the highest for annual horticulture (951 and 1226 mm) reflecting their specific evapotranspiration requirements (Phogat et al., 2018b). Correspondingly, the leaching fraction/recharge flux under almond (87-298 mm) and annual horticulture (100-252 mm) was 3-3.8 times higher than under wine grapes. Besides for wine grapes, a negative annual flux balance was recorded in some years but, the overall average balance was positive over 8 years. These observations are consistent with other studies (Green, 2010; Reynolds, 2010; Phogat et al., 2018b). It is well understood that the contribution of leaching fraction/irrigation return flow from irrigated crops can be a critical driver for the river-buffer hydraulic exchange (e.g., Berens et al., 2009).

![Figure 4. Annual irrigation (mm) and recharge/discharge (mm) in the domain under a) wine grape, b) almond, and 3) annual horticulture (carrot-potato) crops. Positive fluxes are recharge and negative fluxes are discharge from the domain.](image)

3.3. Hydraulic exchange at the river-buffer interface

Hydraulic exchange at this boundary depends on river flow dynamics, rainfall, water uptake by the buffer zone vegetation and irrigated crops, soil evaporation, and shallow groundwater fluctuations near the river. The Gawler River is seasonal and flow depends on the extent of rainfall in the catchment during the winter. Numerous high rainfall/flood events during 2009-2017 resulted in large amounts of recharge as indicated by large peaks in Figure 5. During summer, a large amount of irrigation is applied to crops in the adjoining area, which reverses the flow gradient across the buffer-river boundary. Under such dynamic conditions, aquifer
recharge or discharge may occur depending on the river level (Ghazavi et al., 2012), leading to either a gaining or losing river (Rassam, 2011; Phogat et al., 2017a). However, irrigation-induced flow to the river depends on the water uptake/evapotranspiration pattern of buffer zone vegetation (Phogat et al., 2017a) and on the extent of return flow from the irrigated areas. As shown earlier, the extent of irrigation-induced flow (recharge) is higher for almond and annual horticulture as compared to wine grapes (Figure 4). As a result, the average annual amount of flux exchange at the river interface was nearly twice (1.8-2.1 times) as much under almond and annual horticulture than under wine grapes (Figure 5). Additionally, the high evaporation demand of the buffer zone vegetation may further limit the net discharges to the river. This was probably not met by the low drainage flux under wine grapes. Overall, irrigation induced drainage, groundwater discharge to the stream, evapotranspiration by the buffer zone vegetation and irrigated crops play a key role in defining the extent of exchange between the buffer zone and the river.
Figure 5. Daily flux exchange at the river-buffer interface for almond, wine grape, and annual horticulture (carrot-potato) grown near the Gawler River during 2009-2017.

The impact of different buffer widths on the average annual water exchange at the river interface for different irrigated crops is shown in Figure 6. For wine grapes, the average annual hydraulic balance was negative for the 10-20 m buffers during the simulation period (8 years, 2009-2017), indicating the dominance of flow from the irrigated area to the river system. However, the reverse was observed for buffer widths > 20 m as the evapotranspiration demand of the buffer vegetation governed the water exchange at the river-buffer interface. In the case of almond, however, the overall water balance remained negative (discharge to the river) for a buffer zone up to 65 m due to its 3 times higher irrigation than for wine grapes. Similarly, under annual horticulture (carrot and potato) crops, the overall hydraulic balance was similar.
to almonds and the threshold buffer zone width for equilibrium flow conditions reached at 55 m (Figure. 6). Based on irrigation regime for irrigated crops, different buffer zone widths are required for equilibrium flow conditions at the river-buffer interface.

![Figure 6](image-url)  
**Figure 6.** An average balance of water exchange across the stream-aquifer interface for different buffer widths under a) wine grapes, b) almond, and c) annual horticultural crops.

### 3.4. Impact of a buffer width on solute dynamics

It is well known that irrigation introduces large amount of solutes into the soil as the recycled water contains appreciable amounts of soluble salts (approximately 1200 mg/L; Laurenson et al., 2010). Temporal dynamics of irrigation induced salts in the soil profiles with 30 m buffer widths for wine grapes, almonds, and annual horticulture is shown in Figure. 7. Irrigation induced salts gradually moved downwards as well as laterally, but, the vertical movement was faster. Initially, salts continued to build up in the soil and then migrated to the shallow groundwater (water table at 4 m). It was noted that the irrigation induced salts entered the shallow groundwater within 2 years of irrigation of almond and annual horticulture (Figure. 7), but it took longer for wine grapes. It is noteworthy that after 8 years of simulation, the salts were distributed in variable concentrations throughout the entire domain, including along the river boundary, especially for almond and annual horticulture. The presence of salts along the river boundary indicates that they may have already entered into the river system in low concentrations (< 0.0011 mg/cm³) against the concentration gradient under these crops. In contrast, for wine grapes, salts travelled a far shorter distance than for the other two crops and...
the salt plume remained far from the river front boundary (Figure. 7). Nonetheless, the extent and timing of salts migration into the river system was not clearly visible.

Excessive concentrations of soluble salts as shown on the right hand corner of Figure. 7 for almond and annual horticulture can have an adverse impact on plant growth and transpiration losses. However, this situation was obtained for a 30 m buffer width (Figure. 8), which is an insufficient width to control salts mobilization to the river for these crops. This shows that in the absence of an adequate buffer width, there is a chance of rapid secondary salinization close to the river. In such situations, adopting native vegetation such as river red gums (with a salt tolerance of 30 dS m\(^{-1}\)) or black box trees (salt tolerance of 55 dS m\(^{-1}\)) as buffer (Overton and Jolly, 2004), could help maintain appropriate transpiration services in such situations (Alaghmand et al., 2013, 2014).

**Figure 7.** Temporal dynamics of irrigation induced salts in the soil profile with a 30-m buffer width under a) wine grapes, b) almonds, and c) annual horticulture (carrot-potato) at the indicated times during 2009-2017. The river is situated at the top right corner of the domain.

The frequency, timing, and load of salt pulses entering into the river during 8 years (2009-2017) of simulation under different crops as influenced by buffer zone widths is shown in Figure. 8. It is apparent that for wine grapes, the salts/tracer pulse appeared in the river only for buffer widths < 20 m. Further widening the buffer (30 - 60 m) and extending the simulations to 16 years (2009-2025; data not shown) did not produce any appreciable amount of salts in the river. In contrast, a significant salts/tracer pulse continued to appear in the river for 50 and 60 m buffer for annual horticulture and almond irrigated crops, respectively, during 8 years (2009-2017) of simulation. Further extending the buffer (up to 100 and 110 m) and simulation
time (2009-2025) resulted in a very small amount of tracer salt to appear in the river ($10^{-22}$ to $10^{-6}$ mg) for annual horticulture and almond. Therefore, it is concluded that a buffer width of 20, 50 and 60 m for wine grapes, annual horticulture (carrot-potato) and almonds, respectively, is needed to restrict the subsurface migration of irrigation induced salts/tracer chemicals to the river.

In terms of salt load the same buffer width for different crops released varying pulses in the river (Figure. 8). For example, for a buffer width of 10 m for almond during the 8 years of simulation was approximately twice the salts load than that for annual horticulture, which in turn had 10 times higher loads than wine grapes. However, with increasing buffer width (e.g. 20 m), salts loads were drastically decreased (especially in almonds and annual horticulture), but, the timing of occurrences of salts/tracer pulses in the river are similar irrespective of irrigated crops. Notably, a large solute peak occurred in the river water after approximately 7 years of simulations (August to December, 2016), which transported different amounts of salt to the river at different buffer widths. This time corresponds to the aftermath of a flood event when the water level in the Gawler River reached the soil surface and completely saturated the adjacent riparian zone. Subsequent receding water levels in the river created a steep gradient from the buffer zone to the river, which conveyed a large amount of salts from the saturated zone to the stream. Such observations were also reported in other studies (Bryan et al., 1998; McKergow et al., 2003). Hence, the salts transported to the river may not be associated only with irrigation, but also with the generation of hydraulic gradients, which push the salts laterally into the river. However, the impact of hydraulic gradients gradually dissipates as the buffer width increases, since the buffer zone acts as a barrier in transmitting the hydraulic response between the river and the irrigated area. Overall, it appears that all components of soil, water, crop and climate, play a crucial role and have a different influence on the salt transport to the river and its water quality.
Figure 8. The effect of buffer zone widths (10, 20, 30, 40, 50 and 60 m) on the lateral migration of irrigation induced salts into the river under almond (a to f), annual horticulture (g to k), and wine grapes (l to m) during 2009-2017. The solute flux scale at the vertical axes is different in different figures.

Salts load transported to the river and the residence time of solutes in the soil for different buffer widths and crops are shown in Figure 9. The amount of salts for the 10 m buffer was very similar for almond and annual horticulture and about 40 times higher than for wine grapes. Meanwhile, the salts transported to the river for the 20 m buffer were higher for annual horticulture than for almond. When the buffer width was increased to 60 m, only a small
additional reduction in the salt load was observed (99.9%). Similarly, for annual horticulture, the average annual load of salts was reduced by 92.2% (to 1566 mg) for the 20 m buffer width as compared to the 10 m buffer width. For a 99.9% reduction in the salt load, a 40 m buffer width is needed. Therefore, it is established that maintaining a 20, 60, and 40 m buffer widths for wine grapes, almonds, and annual horticulture can effectively reduce irrigation induced salts/tracers transport to the river by 99.9%.

![Figure 9. The amount of salts (line graph) and their residence times (bars) for different irrigated crops as a function of a buffer width during 2009-2017.](image)

The residence time was twice as long for wine grapes as compared to almonds and annual horticulture for buffers <40 m wide. For example, the residence time for a 10 m buffer was 2.5 months for almonds and annual horticulture, and 5 months for wine grapes (Figure. 9). The residence time for the solute movement to the stream starts increasing exponentially for larger buffer widths (>40 m) and wine grapes. It took about 7 years for irrigation induced solutes, though concentrations were very small, to appear in the stream when the buffer width was 60 m. The residence time for almonds and annual horticulture was very small (<100 days) for buffer widths <30 m, then gradually increased to 450 days for a 70 m, and to much longer values for larger buffer widths. A longer residence time for wine grapes is associated with a relatively low recharge volume, which was unable to generate a sufficient hydraulic gradient to push the salts into the river. Therefore, the extent of irrigation return flow is crucial for the longevity of solutes in the soil system. The knowledge of the time required for the migration of salts to the river helps in devising guidelines for maintaining the river water quality.
4. Implications of the findings

This study demonstrated that for wine grapes, a 20 m buffer width could reduce the load of irrigation salts discharged to the Gawler River down to a negligible level (a 99.9% reduction), whereas for annual horticulture, for a similar salt load reduction, a buffer width of 40 m was needed. On the other hand, for almonds that have much higher irrigation requirement, a buffer zone of 60 m would be needed for a similar reduction. Therefore, the buffer width needs to consider the nature of the production system adjacent to the river system being protected. The decision about an appropriate width of the buffer requires a careful consideration of economic consequences and not just ecological requirements.

This study highlights that considerations of local hydraulic, climate, and soil conditions, as well as local geological heterogeneity, can have a marked impact on the requirement of adequate vegetative buffer along the rivers. The buffer zone guidelines adopted in most of the states (NSW = 40 m, Western Australia = 30 m, Tasmania = 30 m, and Victoria = 60 m) in Australia for maintaining the river water quality are mainly based on overseas studies (Hansen et al., 2010). Therefore, the consideration of the type of irrigated crops grown along the rivers and surface water bodies could have varied impact on the maintenance of the buffer zone and hence the riparian zone guidelines.

5. Conclusions

This study was carried out to understand the extent, frequency and nature of hydraulic and solute dynamics relationship between river and irrigated crops interspersed with a buffer. The calibrated and validated model (HYDRUS-2D) was used to evaluate long-term (8 years) scenarios involving different irrigated crops (wine grapes, almond, and annual horticulture) and varied buffer widths (10 to 110 cm) aimed at quantifying the hydraulic connection between the river and the crop grown and optimising buffer width for controlling solute movement to the river.

Statistical evaluation (correlation coefficient, $ME$, $MAE$, $RMSE$, $t$-test and model efficiency) of the model for water movement showed a fairly good matching between the measured and the simulated water table depths. The results obtained from 8-years of simulations showed that three irrigated crops (wine grapes, almond, and annual horticulture) requiring different annual
irrigation applications have markedly different influence on the overall water and solute movement within the crop-buffer-river ecosystem. Consequently, the average annual flow from the soil to the river was 2.1 and 1.8 times higher under almond and annual horticulture, respectively, compared to wine grapes. For average annual water balance to reach equilibrium, buffer (Eucalyptus spp.) widths of 20, 65, and 55 m were needed for wine grapes, almond, and annual horticultural crops (carrot-potato), respectively.

From the standpoint of migration of irrigation induced soluble salts into the river, the buffer widths of 20, 60, and 40 m were able to reduce the salt load by >99% under wine grapes, almond, and annual horticulture crops, respectively. Notably, existing guidelines do not differentiate between crop types when specifying optimum buffer widths. Further work is required to assess how the seasonal variability in irrigation water quality influences these results, particularly in areas where recycled water is used for irrigation. The modelling challenges and data limitations included the absence of preferential flow, the assumption of similar water stress response functions for the buffer vegetation and irrigated crops, and lack of data for calibration of the model against solute dynamics in the river. Further refinements in the above findings may be achieved by addressing these gaps.

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Appendix 11 Boron risks associated with recycled water irrigation

Table of Contents

1 Introduction ........................................................................................................................................379

2 Methods and materials ..................................................................................................................380
  2.1 Soil chemical and physical properties ..................................................................................380
  2.2 Boron analyses ......................................................................................................................380
  2.3 Boron sorption values ...........................................................................................................381
  2.4 Irrigation water quality ..........................................................................................................382
  2.5 1D solute transport model ...................................................................................................382
      2.5.1 Equilibrium sorption .................................................................................................382
      2.5.2 Non-equilibrium sorption .........................................................................................383

3 Results and discussion ..................................................................................................................384
  3.1 Boron speciation ....................................................................................................................384
  3.2 Native boron concentration in NAP soils ..............................................................................385
      3.2.1 Factors affecting boron availability ...........................................................................385
      3.2.2 Soil solution boron .......................................................................................................385
      3.2.3 Adsorbed soil boron ....................................................................................................388
      3.2.4 Total soil boron ............................................................................................................390
      3.2.5 Boron desorption .........................................................................................................391
      3.2.6 Boron sorption ............................................................................................................393
      3.2.7 Simulated boron behaviour ........................................................................................398

4 Conclusion ....................................................................................................................................406

5 References .......................................................................................................................................407
1 Introduction

Boron (B) is a micronutrient that is required by plants in small quantities (<500 g/ha) (Shorrocks, 1997) and since B is relatively immobile in plants, once utilized in actively growing tissues, it is hardly re-translocated to other parts. Therefore, it is necessary to have a rather continuous source of B available to the plant throughout its growth cycle. Boron is required for the formation of new tissues but not the maintenance of older tissues, so actively growing plants require larger amounts of B than slowly growing or mature plants (Adriano, 2001).

Boron deficiency commonly occurs in sandy soils which have low CEC and OM content, where leaching and heavy cropping have diminished the soil B reserves (Adriano, 2001). Boron toxicity usually is seen in soil of marine sediment, in those soils derived from parent material rich in B, and in arid and semi-arid soils (Adriano, 2001).

Of all plant nutrient elements, the range between deficient and toxic levels of available concentration is smallest for boron (Goldberg, 1997). Small increases due to fertilizer application or via boron in irrigation water and natural variations in boron concentration with soil depth may result in a soil transitioning from deficient to toxic levels or vice versa.

An optimal soil test would measure B capacity, that is, all pools of plant available B including: soluble, organic, and adsorbed. Determination of the total B pool potentially available to plants is necessary for arid and semiarid land soils to evaluate the extent of leaching necessary for reducing B to below toxic levels.

The main factors affecting boron adsorption and plant available boron are soil pH, texture, water content, and temperature.
2 Methods and materials

2.1 Soil chemical and physical properties

Following the soil sampling, soil pH was measured in water ($pH_w$) and in 0.01 M CaCl$_2$ ($pH_{Ca}$). The latter is usually preferred as it is less affected by soil electrolyte concentration and provides more consistent measurements (Minasny et al., 2011). Further details of the analysis can be found in the Task 1 Report (Oliver et al., 2019).

2.2 Boron analyses

The boron concentration in soil solution, and therefore the boron available for plant uptake, is governed by boron adsorption rather than by the solubility of boron containing minerals (Goldberg, 1997). The fraction of B in soil that is available for plant uptake is termed the phytoavailable fraction. Total B is an unreliable measure of the bioavailable fraction in soils and often an extractant, such as water or CaCl$_2$, is used as an index of the phytoavailable fraction (Adriano, 2001). Aitken and McCallum (1988) however, showed that the relationship between B concentrations measured in sunflowers and the hot 0.01M CaCl$_2$ extractable fraction was dependent upon the soil texture. They found a better relationships with the soil solution fraction measured at maximum water holding capacity (MWHC) (termed B in soil solution).

Plants have varying degrees of tolerance to B in soil solution and Adriano (2001) suggests B concentrations <0.5 mg/L in soil solution are probably safe for most plants but many plants may be adversely affected when B levels are in the range of 0.50 to 5.0 mg/L. Threshold concentration ranges for B concentration in soil solution based on Leyshon and Jame (1993) are given in Table 1.

Table 1 Threshold boron concentration in soil solution for various crops and percentage of total samples (0-10, 10-30 and 30-60 cm) within each threshold range. Boron in soil solution determined at maximum water holding capacity. Threshold concentrations from Leyshon and Jame (1993). * data from Gupta et al. (1985).

<table>
<thead>
<tr>
<th>Threshold concentration range for B in soil solution (field capacity basis)</th>
<th>Percentage (number) of soil. Total soils n=82.</th>
<th>Crop within threshold concentration range</th>
</tr>
</thead>
<tbody>
<tr>
<td>(mg/L B)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Very sensitive &lt;0.5</td>
<td>67% (55)</td>
<td>Lemon*, Grapefruit*, Avocado*, Orange*</td>
</tr>
<tr>
<td>Sensitive 0.5-1.0</td>
<td>18% (15)</td>
<td>Fig, Grape, Walnut, Onion, Garlic</td>
</tr>
<tr>
<td>Moderately sensitive 1.0-2.0</td>
<td>5% (4)</td>
<td>Broccoli, Red pepper, Carrot, Potato, Cucumber</td>
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<tr>
<td>Moderately tolerant 2.0-4.0</td>
<td>6% (5)</td>
<td>Lettuce</td>
</tr>
<tr>
<td>Tolerant 4.0-6.0</td>
<td>4% (4)</td>
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</tbody>
</table>
Boron in soil solution was determined at maximum water holding capacity (MWHC) based on McLaughlin et al. (1997). To better represent the soil solution following irrigation, the B in soil solution was measured using a solution with a Cl concentration equivalent to that found in one of the primary irrigation water sources being considered, namely recycled waste water. The Cl concentration in the recycled waste water was 550 mg/L, compared with 4 mg/L in rain water. There was almost a 1:1 relationship between B in soil solution as determined using the high (550 mg/L) and low (4 mg/L) Cl solution (Task 1 Report, Oliver et al., 2019) so only data for the high Cl solutions are discussed below.

A subset of soils were sequentially extracted in duplicate with a high Cl (550 mg/L Cl) solution in a 1:5 ratio. The high Cl solution was selected to represent the highest concentration in the reclaimed water from the DAFF treatment plant. The soils selected had been found to have relatively high B in soil solution and the sequential extractions were performed to determine whether B would continue to come into solution with each successive extraction. The soil solution was shaken on an end-over-end shaker for 24h, then centrifuged at 3000 rpm for 15 mins. The extract was decanted off and a fresh solution was added and the extraction repeated four times. The B extracted was determined by ICP-MS (Task 1 Report, Oliver et al., 2019).

Hot water extraction of boron is a commonly used method for obtaining an index of plant available boron in soils (Cox and Kamprath, 1972; Bingham, 1982). The hot water-soluble procedure measures B capacity in that it extracts B from the organic, adsorbed, and soluble pools of the soil (Offiah and Axley, 1993).

2.3 Boron sorption values

Sorption coefficients (K_d values) were determined for boron (B) using the OECD 106 standard protocol for the adsorption – desorption of chemicals using a batch equilibrium method (OECD/OCDE, 2000). Briefly, the procedure involved weighing 2g of soil into plastic vials in triplicate. There was an initial pre-equilibration for 24 h with 9.5 mL of 0.01M CaCl_2, after which a known volume (0.5 mL) of a spiking solution of B was added to given an initial concentration (starting concentration) of 0.5 mg B/L. The solution was shaken again for 24h on an end-over-end shaker and the concentration remaining in solution was measured using an inductively coupled plasma mass spectrometer (ICP-MS). The soils selected for determining B K_d values were chosen to cover a range of soil textures, soil pH and native B concentrations.

The sorption coefficients (K_d) values were calculated as:

\[ K_d = \frac{\text{Concentration of sorbed chemical (mg/kg soil)}}{\text{Equilibrium concentration in soil solution (mg/L)}} = \frac{C_s}{C_L} \]  

The concentration of the chemical sorbed by the soil was determined as:

\[ C_L = \frac{[\text{Initial concentration (mg/L)}\times\text{volume(L)}][\text{Equilibrium concentration (mg/L)}\times\text{volume(L)}]}{\text{Mass of soil (kg)}} \]  

The equilibrium mass was corrected for native B that came into solution when blank soils were shaken in 1:5 ratio in 0.01M CaCl2 (see Task 1 Report for details, Oliver et al., 2019).
2.4 Irrigation water quality

Boron concentration in recycled DAFF water were obtained from the Task 3 Report (Awad et al., 2019). A summary of the data is provided in Table 2.

Table 2 Boron concentration (mg/L) in Bolivar DAFF irrigation water.

<table>
<thead>
<tr>
<th>Period</th>
<th>No. of observations</th>
<th>5th percentile</th>
<th>50th percentile</th>
<th>95th percentile</th>
</tr>
</thead>
<tbody>
<tr>
<td>2001-2011</td>
<td>122</td>
<td>0.204</td>
<td>0.327</td>
<td>0.529</td>
</tr>
<tr>
<td>2012-2016</td>
<td>57</td>
<td>0.197</td>
<td>0.334</td>
<td>0.529</td>
</tr>
</tbody>
</table>

2.5 1D solute transport model

2.5.1 Equilibrium sorption

Boron concentration in soil solution is considered to be determined mainly by adsorption-desorption reactions (Goldberg, 1997). Boron containing minerals such as tourmaline or hydrated boron minerals are either too insoluble or too soluble to control boron concentration. Main sorption sites for boron in soil are clays, aluminum and iron oxides, magnesium hydroxide, calcite, and organic carbon (Goldberg, 1997). For clay minerals, the order of B adsorption per gram is: kaolinite < montmorillonite < illite (Keren and Mezuman, 1981). Calcium carbonate (CaCO$_3$) acts as an important B adsorbing surface in calcareous soils. Also, addition of CaCO$_3$ increases B fixation by soils because it increases the soil pH (Goldberg and Forster, 1991).

Modelling of equilibrium boron adsorption on oxides, clay minerals, organic matter and other soil solid phases have typically been described using Langmuir and Freundlich adsorption isotherms (Goldberg, 1997; Marzadori et al., 1991). Equilibrium sorption assumes that the adsorption-desorption process is fast relative to fluid movement in soil and thus not time-dependent.

The mathematical expression for the Langmuir equilibrium isotherm is as follows (Zhang and Selim, 2005):

$$S = S_{max} \frac{K_L \times C}{1 + K_L \times C}$$

(3)

where $S$ is the total amount of adsorbed boron ($\mu$g/g), $S_{max}$ represents the sorption maximum that can be related to soil properties ($\mu$g/g), $K_L$ is the Langmuir coefficient which is related to the binding strength (L/mg), and $C$ is the boron concentration in solution (mg/L). The implementation of the Langmuir sorption isotherm in HYDRUS-1D has a slightly different notation (Šimůnek et al., 2008; Mallants et al., 2011):

$$S = K_D \frac{C}{1 + \eta \times C}$$

(4)

where $K_D$ (L/kg) and $\eta$ (L/mg) are empirical or quasi-empirical constants. The parameter $\eta$ in Eq. (4) is equivalent to parameter $K_L$ in Eq (3) while $K_D$ in Eq. (4) is equivalent to $S_{max} \times K_L$.

The Freundlich isotherm is defined as (Šimůnek et al., 2008; Mallants et al., 2011):
\[ S = K_F C^N \]  \hspace{1cm} (5)

where \( K_F \) and \( N \) are the Freundlich parameters.

### 2.5.2 Non-equilibrium sorption

When the adsorption-desorption reaction is partially time dependent, the two-site sorption concept can be implemented (Selim et al., 1977; van Genuchten and Wagenet, 1989). In this case the sorption sites are split into equilibrium sites where sorption is instantaneous (type-1 sites with fast exchange between solid and liquid phase: \( S_{Eq} \) (µg/g)) and sites where sorption is kinetically controlled (type-2 sites with time-dependent sorption: \( S_{Kin} \) (µg/g)). The mathematical expression for the mass balance of the two-site chemical non-equilibrium is (Šimůnek et al., 2008; Mallants et al., 2011):

\[ S_{Tot} = S_{Eq} + S_{Kin} \]  \hspace{1cm} (6)

\[ S_{Eq} = f S_{Tot} \]  \hspace{1cm} (7)

\[
\frac{\partial S_{Kin}}{\partial t} = \alpha \left[ (1 - f) K_D \frac{C}{1 + \eta C} - S_{Kin} \right] \]  \hspace{1cm} (8)

where \( S_{Tot} \) represent the total sorbed phase concentration (µg/g), \( f \) (-) is the fraction of equilibrium sorption sites (defined by Eq. (7)), \( \alpha \) is a first-order rate constant (h\(^{-1}\)), and \( K_D \) and \( \eta \) are as defined previously. In Eq. (8) we have assumed that equilibrium sorption is represented by a Langmuir type isotherm, although linear and Freundlich isotherms can also be invoked.
3 Results and discussion

3.1 Boron speciation

Boron chemistry in soil is very simple: it does not display redox reactions and is not volatile. Dissolved boron in soil pore water is present only in the +3 valence state (Figure 1). The dominant boron species in soils at low pH values is the neutral and weak boric acid, B(OH)₃. As the soil pH increases, boric acid forms the borate anion by accepting a hydroxyl ion: as a result, the proportion of the borate anion, B(OH)₄⁻, increases (Figure 1b). The ratio of these two boron species depends on the first dissociation constant of boric acid, $K_a$. The fresh water $K_a$ value at temperature 25 °C is equal to $5.8 \times 10^{-10}$ mol/L. The corresponding $pK_a$ value = 9.24 ($pK_a = - \log(K_a)$). As a result, at pH < 9.2, dissolved boron predominantly exists in the form of an uncharged oxyanion (B(OH)₃) while at pH > 9.2, it is mostly present as B(OH)₄⁻ (Figure 1).

The boron $pK_a$ value is known to be dependent on salinity and temperature. A closed-form equation was developed by Dickson (1990) to quantify the salinity and temperature dependency. We implemented this equation to calculate the $pK_a$ values for i) a temperature range from 5 to 45°C and, ii) a low (2042 mg/L, based on a maximum measured soil EC was 3712 μS/cm) and high salinity (5000 mg/L). Note that the lowest salinity level S in the Dickson model was 5000 mg/L, hence effects of the low salinity will be immaterial. Based on these calculated $pK_a$ values, the speciation model Phreeqc was run to test the effect on the B(OH)₃ and B(OH)₄⁻ species (Figure 1b). Temperature effects on the speciation are noticeable, but small. We conclude that the standard $pK_a$ value at 25°C is suitable for the NAP conditions.

![Figure 1 Eh–pH predominance diagram of boron at 25 °C, 1.013 bars and activity of B = 10⁻⁶ calculated with The Geochemist’s Workbench™ (Bethke et al., 2019) (left). Distribution of aqueous boron species versus pH in soil water at low EC (ionic strength I = 2.714×10⁻² mol/kgw between pH 7-9) and high EC (5 g/L salinity) (right). Boron speciation calculations based on Phreeqc (Parkhurst and Appelo, 2013).](image)

Boron adsorption in soil is predominantly on oxides, clay minerals, calcite, and organic matter (Goldberg et al., 1993). The adsorption of boron in US soils was shown to be pH dependent with a maximum sorption at pH 6 to 8 for soil rich in Al-oxides and between pH 7 to 9 for soils with predominantly Fe-oxides (Goldberg and Glaubig, 1985; Goldberg et al., 1993) (Figure 2). At low pH, B sorption is low and dominated by the neutral species B(OH)₃. As the pH increases, the borate ion,
B(OH)$_4^-$ becomes the most abundant species characterized by an increase in sorption (Goldberg and Glaubig, 1986; Adriano, 2001). However, further increases in pH (i.e. pH > 9) result in increased OH$^-$ concentration relative to B(OH)$_4^-$ causing B adsorption to decrease due to competition with OH$^-$ for sorption sites (Goldberg and Glaubig, 1986; Adriano, 2001).

Boron adsorption for soils from the Northern Adelaide Plains will be discussed in the next section.

![Figure 2 Adsorption of boron versus pH for a variety of soil](from Goldberg et al., 2000).

### 3.2 Native boron concentration in NAP soils

#### 3.2.1 Factors affecting boron availability

Boron availability in soils is mainly affected by pH: as pH increases, boron adsorption increases and therefore its availability to plants decreases. As shown in Figure 2, boron adsorption increases as function of soil pH in the range 3 to 9, and decreases in the pH range 10-11.5.

#### 3.2.2 Soil solution boron

Of all the soils assessed (surface layers 0-10, 10-30 and 30-60 cm), boron in soil solution was low (<0.5 mg/L B) for 72% of soils, while 13% of soils had >1 mg/L and 8% had >2 mg/L boron in soil solution (Task 1 Report, Oliver et al., 2019). Comparison of these boron levels with threshold values above which crops become sensitive to B indicates that the native B in some of the soils may already be at concentrations that are limiting or toxic to crop growth (Table 1).
The distribution of B in soil solution at the three sampling depths for the soils sampled across the NAP region is shown in Figure 3 (and in Figure 4 expressed per kg soil) and listed in Table 3. Hard red brown soils display the overall highest concentrations, where for the remaining three soils boron concentrations are similar. Also, hard red brown soils have the highest relative variability in boron as expressed through the coefficient of variation, CV (Table 3).

Boron concentration for all soils increases with depth. This may be due to several reasons, including higher organic matter in the top soil layer providing significant sorption capacity, higher concentrations of boron minerals at greater soil depth (inferred from total B versus depth, Figure 4), and leaching of boron under natural rainfall conditions. The mean boron concentration versus depth data will be used as initial solute conditions for the simulations with HYDRUS-1D; as the model domain is 2 m deep, the measured concentration at the 0.45 m depth is used throughout the remaining soil depth as initial concentration.

Figure 3 Boron in soil solution for soils of the Northern Adelaide Plains (data from Task 1).
Table 3 Boron in soil solution as extracted using 550 mg/L chloride solution. Stdev = standard deviation, CV = coefficient of variation (=100x(Stdev/mean)), N = number of observations. Data expressed as mg/kg are available in Figure 4 (data from Task 2).

<table>
<thead>
<tr>
<th>Soil depth (cm)</th>
<th>Min (mg/L)</th>
<th>Max (mg/L)</th>
<th>Mean (mg/L)</th>
<th>Median (mg/L)</th>
<th>Stdev (mg/L)</th>
<th>CV (%)</th>
<th>N</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Hard red brown soil</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-10</td>
<td>0.12</td>
<td>0.77</td>
<td>0.29</td>
<td>0.23</td>
<td>0.21</td>
<td>70.7</td>
<td>11</td>
</tr>
<tr>
<td>10-30</td>
<td>0.12</td>
<td>9.1</td>
<td>1.3</td>
<td>0.35</td>
<td>2.63</td>
<td>201.8</td>
<td>12</td>
</tr>
<tr>
<td>30-60</td>
<td>0.06</td>
<td>11</td>
<td>2.07</td>
<td>0.80</td>
<td>2.99</td>
<td>142.2</td>
<td>16</td>
</tr>
<tr>
<td><strong>Deep uniform to gradational soil</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-10</td>
<td>0.12</td>
<td>0.29</td>
<td>0.21</td>
<td>0.21</td>
<td>0.12</td>
<td>56.9</td>
<td>2</td>
</tr>
<tr>
<td>10-30</td>
<td>0.19</td>
<td>0.60</td>
<td>0.34</td>
<td>0.27</td>
<td>0.18</td>
<td>54.1</td>
<td>4</td>
</tr>
<tr>
<td>30-60</td>
<td>0.23</td>
<td>1.21</td>
<td>0.63</td>
<td>0.65</td>
<td>0.39</td>
<td>61.8</td>
<td>5</td>
</tr>
<tr>
<td><strong>Sand over clay soil</strong></td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>0-10</td>
<td>0.09</td>
<td>0.11</td>
<td>0.10</td>
<td>0.11</td>
<td>0.01</td>
<td>9.8</td>
<td>5</td>
</tr>
<tr>
<td>10-30</td>
<td>0.16</td>
<td>0.26</td>
<td>0.21</td>
<td>0.22</td>
<td>0.05</td>
<td>22.2</td>
<td>4</td>
</tr>
<tr>
<td>30-60</td>
<td>0.33</td>
<td>0.56</td>
<td>0.44</td>
<td>0.44</td>
<td>0.13</td>
<td>29.3</td>
<td>4</td>
</tr>
<tr>
<td><strong>Calcareous soil</strong></td>
<td></td>
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</tr>
<tr>
<td>0-10</td>
<td>0.14</td>
<td>0.31</td>
<td>0.24</td>
<td>0.27</td>
<td>0.08</td>
<td>31.8</td>
<td>5</td>
</tr>
<tr>
<td>10-30</td>
<td>0.21</td>
<td>0.38</td>
<td>0.3</td>
<td>0.3</td>
<td>0.07</td>
<td>24.4</td>
<td>6</td>
</tr>
<tr>
<td>30-60</td>
<td>0.26</td>
<td>2.86</td>
<td>1.24</td>
<td>0.75</td>
<td>1.11</td>
<td>90.1</td>
<td>6</td>
</tr>
</tbody>
</table>
3.2.3 Adsorbed soil boron

The concentration of adsorbed boron was determined in two ways. The first method uses the hot water extraction of boron in 0.01M CaCl₂, commonly used to measure B extracted from the adsorbed pools (organic, clays) and soluble pools of the soil (Offiah and Axley, 1993). The second method uses the batch method to determine $K_d$, in which the absorbed concentration is one of the measured parameters (see higher).

Adsorbed boron determined with the hot water extraction is shown in Figure 5 for hard red brown and deep uniform to gradational soil. Hard red brown soil have a maximum of adsorbed boron at about 40 cm depth, whereas the deep uniform to gradational soil display increasing adsorbed boron concentrations with depth. Boron behaviour in hard red brown soils is likely related to the depth distribution of the clay fraction; a typical clay fraction is 14% (0-10 cm), 36% (10-30 cm), 55% (30-60 cm), 37% (60-90 cm), and 29% (90-120 cm). In the deep uniform to gradational soil, clay content...
more or less continues to increase with depth; a typical profile is 12% (0-10 cm), 4% (10-30 cm), 16% (30-60 cm), 27% (60-90 cm), and 27% (90-120 cm).

Figure 5 Adsorbed boron for soils of the Northern Adelaide Plains based on hot water extraction (data from Task 1). Hard red brown based on data from soil profile NAP3, NAP4, NAP6, and NAP7. Deep uniform to gradational from soil profile NAP1, NP2, NAP5.

Table 4 Adsorbed boron using hot water extract. Stdev = standard deviation, CV = coefficient of variation (100x(Stdev/mean)), N = number of observations.

<table>
<thead>
<tr>
<th>Soil depth (cm)</th>
<th>Min (mg/kg)</th>
<th>Max (mg/kg)</th>
<th>Mean (mg/kg)</th>
<th>N</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Hard red brown soil</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-10</td>
<td>1.04</td>
<td>2.88</td>
<td>1.74</td>
<td>4</td>
</tr>
<tr>
<td>10-30</td>
<td>1.00</td>
<td>39.82</td>
<td>12.88</td>
<td>4</td>
</tr>
<tr>
<td>30-60</td>
<td>2.10</td>
<td>28.46</td>
<td>14.25</td>
<td>4</td>
</tr>
<tr>
<td>60-100</td>
<td>1.78</td>
<td>27.06</td>
<td>10.57</td>
<td>6</td>
</tr>
<tr>
<td>100-200</td>
<td>2.98</td>
<td>20.47</td>
<td>11.36</td>
<td>4</td>
</tr>
<tr>
<td><strong>Deep uniform to gradational soil</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0-10</td>
<td>0.72</td>
<td>2.09</td>
<td>1.30</td>
<td>3</td>
</tr>
<tr>
<td>10-30</td>
<td>0.76</td>
<td>2.48</td>
<td>1.45</td>
<td>3</td>
</tr>
<tr>
<td>30-60</td>
<td>1.4</td>
<td>7.33</td>
<td>2.96</td>
<td>5</td>
</tr>
<tr>
<td>60-100</td>
<td>2.1</td>
<td>17.23</td>
<td>7.17</td>
<td>3</td>
</tr>
<tr>
<td>100-200</td>
<td>3.67</td>
<td>24.6</td>
<td>12.08</td>
<td>5</td>
</tr>
</tbody>
</table>
3.2.4 Total soil boron

The aqua regia digestion method (US-EPA 3050 (1996) or ISO standard 11466 (1995)) is considered effective for measuring “total” trace element in soils and is usually used to give an estimate of the maximum element availability to plants. Depth distribution of total soil boron is shown in Figure 6 for all four soils. With hard red browns showing the overall highest concentrations, these soils could potentially cause suboptimal crop production for boron sensitive crops. Note that the total soil boron determined with the aqua regia digestion is sometimes lower than that determined with the hot water extraction shown in Figure 5. This is because measurements for those two measurements were based on samples from slightly different depths; given the vertical variability in clay, organic matter, mineralogy and other properties, vertical variability in total boron is expected to be considerable. It is therefore best to consider both the aqua regia digestion and the hot water extract together to estimate the total boron concentration in this study.

Figure 6 Total boron in soils of the Northern Adelaide Plains (microwave soil digest using reverse aqua regia) (data from Task 1).
Total boron concentration in hard red brown was shown to be correlated with clay% and pH (Figure 7); for other soils such correlation was not evident (likely due to the limited number of data points).

3.2.5 Boron desorption

Sequential leaching tests over a period of 96 hours provided data on time-dependent boron desorption which were used to derive the first-order rate coefficient $\alpha$ (h$^{-1}$). This analysis was undertaken only for hard red brown soil, as this is the soil with the highest concentrations and thus presents the highest risk. Figure 8a and b show the desorption curves for hard red brown soil depicted as liquid phase concentration (mg/L) versus time from which the cumulative desorbed boron concentration (mg/kg) was derived. In order to derive the first-order mass transfer coefficient $\alpha$, the cumulative desorbed boron data were re-arranged based on the following form of the first-order kinetic expression (Pavlatou and Polyzopoulos, 1988):

*Figure 7 Total boron as function of calcite, clay, and pH (data from Task 1).*
\[ \ln(S_{\text{max}} - S) = \ln S_{\text{max}} - \alpha t \] (9)

where \(S_{\text{max}}\) is the total sorption capacity of the soil (mg/kg), \(S\) is the sorbed amount (mg/kg) at time \(t\) (h). Values for \(S_{\text{max}}\) were put equal to the total boron concentration obtained by microwave soil digest using reverse aqua regia. Least squares fitting of Eq. (9) with \(S_{\text{max}}\) fixed at independently measured values yielded the first-order rate coefficient \(\alpha\) (Figure 8b and d). Table 5 provides a summary of parameters of Eq. (9).

The fraction of equilibrium sites, \(f\), was calculated according to Eq. (7) from values of \(S_{\text{max}}\) and the total desorbed boron at the end of the desorption tests, considered to be equivalent to \(S_{Eq}\). This assumes that the desorbed boron in the first 96 hrs is readily available for desorption and that it

Figure 8 Desorption data (a, c) and fitted first-order kinetic model (b, d). The desorption includes instantaneous (left axis) and cumulative values (right axis). Fitted parameters for models in (d) are given in Table 5.
provides a reasonable estimation of the boron on the equilibrium sites. Although the desorption curves have not yet reached a steady-state (Figure 8a, c), considering the very long simulation times (several tens of years) considered here, the desorption data represented here can be considered to only represent relatively rapidly released boron. The second approach assumes that the total desorbed concentration obtained through the hot water extraction is a good estimator of \( S_{\text{eq}} \). Both approaches to estimate \( f \) are included in Table 5.

The inherent limitation in the data is that the short time scale involved cannot provide accurate reaction parameters for the long-term release of boron. For this reason the simulated boron releases are considered a sensitivity analysis only, and are not aimed at accurately predicting the boron behaviour in the soil profile under intense irrigation with recycled water.

### Table 5 Fitted first-order kinetic parameter \( S_{\text{max}} \) for data shown in Figure 8. Hard red brown soils (desorbed after 96 hrs). \( S_{\text{max}} \) = total boron from microwave soil digest; \( f = S_1/S_{\text{max}} \), where \( S_1 \) is cumulative desorbed boron at \( t = 96 \) hrs or total desorption from hot water extraction; \( r \) = correlation coefficient for linear regression.

<table>
<thead>
<tr>
<th>Soil depth</th>
<th>Reaction rate constant ( \alpha ) (h(^{-1}))</th>
<th>( \ln(S_{\text{max}}) )</th>
<th>Correlation coefficient ( r )</th>
<th>Fraction of ( S_1 ) sites, ( f ) (-)</th>
<th>Fraction of ( S_1 ) sites, ( f ) (-) [hot water extraction]</th>
</tr>
</thead>
<tbody>
<tr>
<td>CL014 20 cm</td>
<td>0.0070</td>
<td>4.177</td>
<td>0.958</td>
<td>0.467</td>
<td>0.0266</td>
</tr>
<tr>
<td>CL014 40 cm</td>
<td>0.01059</td>
<td>4.177</td>
<td>0.924</td>
<td>0.743</td>
<td>0.197</td>
</tr>
<tr>
<td>NAP06 20 cm</td>
<td>0.00152</td>
<td>4.177</td>
<td>0.962</td>
<td>0.605</td>
<td>0.610</td>
</tr>
<tr>
<td>NAP06 45 cm</td>
<td>0.0105</td>
<td>5.289</td>
<td>0.938</td>
<td>0.605</td>
<td>0.457</td>
</tr>
<tr>
<td>NAP07 45 cm</td>
<td>0.0129</td>
<td>3.663</td>
<td>0.977</td>
<td>0.692</td>
<td>0.366</td>
</tr>
<tr>
<td>NAP13 45 cm</td>
<td>0.0128</td>
<td>3.200</td>
<td>0.987</td>
<td>0.692</td>
<td>0.580</td>
</tr>
<tr>
<td>NAP15 45 cm</td>
<td>0.0164</td>
<td>3.610</td>
<td>0.998</td>
<td>0.797</td>
<td>0.385</td>
</tr>
<tr>
<td>NAP20 45 cm</td>
<td>0.00937</td>
<td>3.761</td>
<td>0.966</td>
<td>0.569</td>
<td>0.331</td>
</tr>
</tbody>
</table>

### 3.2.6 Boron sorption

Three types of sorption isotherm were derived for boron sorption, based on hard red brown soil: linear, Freundlich, and Langmuir.

The Langmuir parameters \( S_{\text{max}} \) and \( K_l \) were obtained by fitting Eq. (3) to the isotherm data from Figure 9. The following data points were taken into consideration: equilibrium boron concentrations in liquid and solid phase from batch tests, and total desorbed boron concentration using the hot water extraction with 0.01 M CaCl\(_2\) solution. The latter is considered to provide an estimate of the boron adsorbed on clay minerals and organic carbon and boron in the soluble pools (Offiah and Axley, 1993). In estimating parameters of Eq. (3) we assumed the hot water extraction data represents \( S_{\text{max}} \), which was then fixed for fitting the remaining parameter, i.e. with \( K_l \) the only fitting parameter. Fitting by eye was done with results shown in Table 6; because only very few data points were available, fitting by eye is not worse than automated error minimization.
Results from the parameter estimation must be treated with care. Indeed, because the use of adsorption isotherms is basically a curve fitting exercise, the fitting parameters are only valid for the conditions under which the experiment was conducted. Therefore, prediction of B adsorption for conditions beyond those of the experiment will be highly unreliable, especially if this involves changes of soil solution B concentration, pH, and ionic strength (Goldberg, 1997).

For hard red brown soils four data sets were considered for estimation of the Langmuir isotherms. The data originates from four soil profiles (NAP4, NAP6, NAP7, and NAP13) at three depths (0-10, 10-30, and 30-60 cm). The liquid phase/solid phase data were grouped according to the main soil sorbing material, i.e. clay and calcite. The four data groups also have a distinctly different pH. The first group (0-10 cm for NAP 4 and 10-30 cm for NAP7) has a pH between 5.9 and 6.3 with an equilibrium boron pore-water concentration of 0.54-0.57 mg/L (Figure 9). Both soil layers have a
relatively low clay percentage (8-14%) and low levels of calcite (0.1%). The fitted Langmuir isotherm (“Langmuir 1”) is shown in Figure 9, with parameters listed in Table 6.

The second data group (30-60 cm from NAP13) has a pH of 8.4 and equilibrium boron concentrations of 1 mg/L. This soil layer has high clay content (44%) but low calcite (0.1%). Parameters for the “Langmuir 2” model are provided in Table 6 while Figure 9 shows data and fitted isotherm.

The third data group (30-60 cm from NAP7) has a pH of 8.9, the boron concentration in solution is 3.1 mg/L, and the percentage clay and calcite are 33 and 1.1%, respectively. Estimated parameters for the “Langmuir 3” model are given in Table 6 while data and model are shown in Figure 9.

The fourth data group (10-30 cm from NAP6) has a pH of 8.9, pore-water boron concentration of 5.8 mg/L, 27% clay and 3.2% calcite. This group has the overall highest value for S_{max} (Table 6) with corresponding isotherm “Langmuir4” shown in Figure 9.

Isotherm parameters were not determined for the other soil groups mainly for two reasons: i) there is no desorption data for those soils hence there are no first-order kinetic parameter values, ii) hard red brown soil has the highest risk of boron toxicity, hence we focus on this soil group.

Table 6 Parameters of the Langmuir sorption isotherms for boron (hard red brown soil).

<table>
<thead>
<tr>
<th>Soil depth (cm)</th>
<th>Soil description</th>
<th>Langmuir parameters</th>
<th>Model name</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-10 (NAP4)</td>
<td>14% clay; 0.1% calcite</td>
<td>1.52 1.5</td>
<td>Langmuir 1</td>
</tr>
<tr>
<td>10-30 (NAP7)</td>
<td>8% clay; 0.1% calcite</td>
<td>12.14 0.35</td>
<td>Langmuir 2</td>
</tr>
<tr>
<td>30-60 (NAP13)</td>
<td>44% clay; 0.1% calcite</td>
<td>21.37 0.18</td>
<td>Langmuir 3</td>
</tr>
<tr>
<td>30-60 (NAP7)</td>
<td>33% clay; 1.1% calcite</td>
<td>65.23 0.053</td>
<td>Langmuir 4</td>
</tr>
</tbody>
</table>

The sorption maximum S_{max} is clearly related to the soil properties % clay and % calcite: the lowest % of clay and calcite produced the lowest value for S_{max} (1.52, see Table 6), while the soil with the highest % calcite (3.2) and high % clay (27) had the highest S_{max} (65.23, see Table 6). Intermediate values for S_{max} exist for the soils with intermediate % calcite (1.1%) and high % clay (33%) or high % clay (44) but low calcite (0.1%). High calcite seems to dominate over high clay content; this is explained by the higher pH (8.9) when calcite is high (1.1-3.2%). As was shown in the boron speciation diagram (Figure 1), the boron species B(OH)_{3} and B(OH)_{4} are present in about equal fractions with boron sorption being close to or at its maximum (also see Figure 2).

For the purpose of simulating boron behaviour in soil two hypothetical soil profiles were composed, each consisting of a set of the previously defined Langmuir models (Table 7). Type-1 profile is build around Langmuir model 1 and 3 has small S_{max} values from 0 to 30 cm, then S_{max} increases to medium values for the remainder of the profile. Type-2 profile is based on Langmuir model 1, 2, and...
4 and has a maximum $S_{\text{max}}$ from 15-30 cm, representing the effect of a higher clay percentage. For both profiles the Langmuir parameters have also been recalculated according to Eq. (4), which is the default model used in HYDRUS-1D (Table 7).

Table 7 HYDRUS-1D input parameters for Langmuir sorption isotherms (hard red brown soil).

<table>
<thead>
<tr>
<th>Soil type</th>
<th>Soil depth (cm)</th>
<th>Soil material number</th>
<th>Langmuir parameters (Eq. 3)</th>
<th>Langmuir parameters Eq. (4)</th>
<th>Model name</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>$S_{\text{max}}$ (µg/g)</td>
<td>$K_L$ (L/mg)</td>
<td>$\eta$ (L/mg)</td>
</tr>
<tr>
<td>Hard red brown Type-1</td>
<td>0-15</td>
<td>1</td>
<td>1.52</td>
<td>1.5</td>
<td>1.5</td>
</tr>
<tr>
<td></td>
<td>15-30</td>
<td>2</td>
<td>1.52</td>
<td>1.5</td>
<td>1.5</td>
</tr>
<tr>
<td></td>
<td>30-60</td>
<td>3</td>
<td>21.37</td>
<td>0.18</td>
<td>0.18</td>
</tr>
<tr>
<td></td>
<td>60-100</td>
<td>4</td>
<td>21.37</td>
<td>0.18</td>
<td>0.18</td>
</tr>
<tr>
<td></td>
<td>100-200</td>
<td>5</td>
<td>21.37</td>
<td>0.18</td>
<td>0.18</td>
</tr>
<tr>
<td>Hard red brown Type-2</td>
<td>0-15</td>
<td>1</td>
<td>1.52</td>
<td>1.5</td>
<td>1.5</td>
</tr>
<tr>
<td></td>
<td>15-30</td>
<td>2</td>
<td>65.23</td>
<td>0.053</td>
<td>0.053</td>
</tr>
<tr>
<td></td>
<td>30-60</td>
<td>3</td>
<td>12.14</td>
<td>0.35</td>
<td>0.35</td>
</tr>
<tr>
<td></td>
<td>60-100</td>
<td>4</td>
<td>12.14</td>
<td>0.35</td>
<td>0.35</td>
</tr>
<tr>
<td></td>
<td>100-200</td>
<td>5</td>
<td>12.14</td>
<td>0.35</td>
<td>0.35</td>
</tr>
</tbody>
</table>

Fitted parameters for the Freundlich isotherm (Eq. 5) for hard red brown soil are listed in Table 8. The same data groups were used as with the Langmuir isotherms.

Table 8 Parameters of the Freundlich sorption isotherms for boron (hard red brown soil).

<table>
<thead>
<tr>
<th>Soil depth (cm)</th>
<th>Soil description: Clay/calcite</th>
<th>Freundlich parameters Eq. (5)</th>
<th>Model name</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>$N$ (µg/g)</td>
<td>$K_F$ (L/kg)</td>
</tr>
<tr>
<td>0-10 (NAP4)</td>
<td>14% clay; 0.1% calcite</td>
<td>0.40</td>
<td>1.1</td>
</tr>
<tr>
<td>10-30 (NAP7)</td>
<td>8% clay; 0.1% calcite</td>
<td></td>
<td></td>
</tr>
<tr>
<td>30-60 (NAP13)</td>
<td>44% clay; 0.1% calcite</td>
<td>0.60</td>
<td>3.1</td>
</tr>
<tr>
<td>30-60 (NAP7)</td>
<td>33% clay; 1.1% calcite</td>
<td>0.57</td>
<td>4</td>
</tr>
<tr>
<td>10-30 (NAP6)</td>
<td>27% clay; 3.2% calcite</td>
<td>0.64</td>
<td>5</td>
</tr>
</tbody>
</table>
Finally, also a linear isotherm was assumed with $K_d$ values list in Table 9. These values were derived based on the methodology described in section 4.7.2.3.

Table 9 Parameters of the linear sorption isotherms for boron (hard red brown soil). # number of observations

<table>
<thead>
<tr>
<th>Soil depth (cm)</th>
<th>Soil description: Clay/calcite</th>
<th>$K_d$ (L/kg)</th>
<th>Model name</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-10 (NAP4)</td>
<td>14% clay; 0.1% calcite, 8% clay; 0.1% calcite</td>
<td>1.281 (#6)</td>
<td>Linear 1</td>
</tr>
<tr>
<td>10-30 (NAP7)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>30-60 (NAP13)</td>
<td>44% clay; 0.1% calcite</td>
<td>3.231 (#3)</td>
<td>Linear 2</td>
</tr>
<tr>
<td>30-60 (NAP7)</td>
<td>33% clay; 1.1 % calcite</td>
<td>2.490 (#3)</td>
<td>Linear 3</td>
</tr>
<tr>
<td>10-30 (NAP6)</td>
<td>27% clay; 3.2% calcite</td>
<td>2.614 (#3)</td>
<td>Linear 4</td>
</tr>
</tbody>
</table>

The non-equilibrium parameters $\alpha$ and $f$ as used in HYDRUS-1D for the two-site sorption model are listed in Table 9. Two cases are considered, which differ only in the values of the fraction of S1 sites, $f$. The first method derived $f$ from $S_{max}$ based on the amount of boron desorbed after 96 hrs; while the second method derived $f$ from $S_{max}$ based on the hot water extract (Section 4.7.2.2). As $S_{max}$ values for the second method are larger than those of the first method, its $f$ values are slightly smaller.

Table 10 Two-site sorption model parameters used in Hydrus-1D model. Fraction of S1 sites, $f$, calculated as $f = S_1/S_{max}$, where $S_1$ is cumulative desorbed boron at $t = 96$ hrs (method 1) or desorbed from hot water extract (method 2).

<table>
<thead>
<tr>
<th>Soil type</th>
<th>Soil depth</th>
<th>Reaction rate constant $\alpha$ (h$^{-1}$)</th>
<th>Reaction rate constant $\alpha$ (d$^{-1}$)</th>
<th>Fraction of S1 sites, $f$ (-)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hard red brown – method 1 (S1 desorption after 96 hrs)</td>
<td>0-15 cm</td>
<td>0.00426</td>
<td>0.102</td>
<td>0.536</td>
</tr>
<tr>
<td></td>
<td>15-30 cm</td>
<td>0.00426</td>
<td>0.102</td>
<td>0.743</td>
</tr>
<tr>
<td></td>
<td>30-60 cm</td>
<td>0.0129</td>
<td>0.309</td>
<td>0.727</td>
</tr>
<tr>
<td></td>
<td>60-100 cm</td>
<td>0.0129</td>
<td>0.309</td>
<td>0.727</td>
</tr>
<tr>
<td></td>
<td>100-200 cm</td>
<td>0.0129</td>
<td>0.309</td>
<td>0.727</td>
</tr>
<tr>
<td>Hard red brown – method 2 (desorbed from hot water extract)</td>
<td>0-15 cm</td>
<td>0.00426</td>
<td>0.102</td>
<td>0.318</td>
</tr>
<tr>
<td></td>
<td>15-30 cm</td>
<td>0.00426</td>
<td>0.102</td>
<td>0.318</td>
</tr>
<tr>
<td></td>
<td>30-60 cm</td>
<td>0.0129</td>
<td>0.309</td>
<td>0.415</td>
</tr>
<tr>
<td></td>
<td>60-100 cm</td>
<td>0.0129</td>
<td>0.309</td>
<td>0.415</td>
</tr>
<tr>
<td></td>
<td>100-200 cm</td>
<td>0.0129</td>
<td>0.309</td>
<td>0.415</td>
</tr>
</tbody>
</table>
A final step in setting up the boron sorption model is defining the initial boron concentrations in the soil profile. In HYDRUS-1D, both the liquid phase concentration (mass of solute per volume of water) and the adsorbed concentration (mass of solute per mass of soil) can be defined. Internally in the code, the total boron concentration, \( S_T \) (µg/cm³ soil), is then defined as:

\[
S_T = \Theta C + \rho_b S
\]

where \( \Theta \) is soil water content (cm³/cm³), \( C \) is liquid phase concentration (mg/L or µg/cm³), \( \rho_b \) is solid density (g/cm³), and \( S \) is sorbed concentration (µg/g). Liquid phase concentration was measured by extraction at maximum water holding capacity, where the latter is defined as the water content at -5 kPa (McLaughin et al., 1997) (Task 1 Report, Oliver et al., 2019). These water content values were obtained from the water retention measurements (Section 3.2). The \( S \) parameter was based on the hot water extraction method, while bulk density was separately measured as part of a comprehensive set of soil physical measurements (Section 3.1). The relevant values for \( C \) and \( S \) are given in Table 11.

<table>
<thead>
<tr>
<th>Soil type</th>
<th>Soil depth (cm)</th>
<th>Soil material number</th>
<th>Initial liquid-phase concentration (µg/cm³)</th>
<th>Initial sorbed concentration (µg/g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hard red brown – Type 1</td>
<td>0-15</td>
<td>1</td>
<td>0.29**</td>
<td>1.74***</td>
</tr>
<tr>
<td></td>
<td>15-30</td>
<td>2</td>
<td>1.3**</td>
<td>12.88***</td>
</tr>
<tr>
<td></td>
<td>30-60</td>
<td>3</td>
<td>2.07**</td>
<td>14.25***</td>
</tr>
<tr>
<td></td>
<td>60-100</td>
<td>4</td>
<td>2.07**</td>
<td>10.57***</td>
</tr>
<tr>
<td></td>
<td>100-200</td>
<td>5</td>
<td>2.07**</td>
<td>11.36***</td>
</tr>
<tr>
<td>Hard red brown – Type 2</td>
<td>Soil depth (cm)</td>
<td>Soil material number</td>
<td>Initial liquid-phase concentration (µg/cm³)*</td>
<td>Initial sorbed concentration (µg/g)#</td>
</tr>
<tr>
<td></td>
<td>0-15</td>
<td>1</td>
<td>0.77</td>
<td>1.50</td>
</tr>
<tr>
<td></td>
<td>15-30</td>
<td>2</td>
<td>0.90</td>
<td>14.28</td>
</tr>
<tr>
<td></td>
<td>30-60</td>
<td>3</td>
<td>3.8</td>
<td>28.46</td>
</tr>
<tr>
<td></td>
<td>60-100</td>
<td>4</td>
<td>3.8</td>
<td>19.24</td>
</tr>
<tr>
<td></td>
<td>100-200</td>
<td>5</td>
<td>3.8</td>
<td>20.47</td>
</tr>
</tbody>
</table>

### 3.2.7 Simulated boron behaviour

Simulation of boron behaviour in soil was undertaken in a similar way as for the salinity and sodicity simulations, i.e. starting with a warming up period to initialise the soil (from 1970-2017), followed by
the 32 year of irrigation with recycled water (2018-2050). During the warming up period, the only boron added to the soil was via the rainwater, with an average boron concentration of 0.05 mg/L (Crosbie et al., 2012). From 2018 onwards, irrigation water with an average boron concentration of 0.33 mg/L is added to the soil. Pasture is as crop, with water requirements as previously estimated in Appendix 4.

The distribution of materials (soil horizons), initial solid phase solute concentration and the depth locations of observations points are shown in Figure 10.

The two-site non-equilibrium sorption model is selected with parameter estimated in the previous sections. Sorption on the equilibrium site is described by means of the Langmuir isotherm. Given the uncertainty around the kinetic mass exchange parameter $\alpha$, the simulations of boron behaviour following irrigation with recycled water are to be considered primarily as sensitivity analysis. Initial simulations with the derived $\alpha$ resulted in an unrealistic boron behaviour, with most of the adsorbed boron being released in the first few years after the start of the simulations. As mentioned in Section 4.7.3.2.5, the duration of the leaching test was too short to be useful for deriving reaction parameters representative of long-term, slow release processes. Therefore, the $\alpha$ was arbitrarily decreased by a $10^{-4}$ and $10^{-5}$ to mimic very slow release of boron from the kinetically controlled sorption sites. The boron behaviour is simulated first considering the adsorbed boron concentrations of profile NAP7 (Table 11). Subsequent sensitivity analysis will be carried out to analyse the effect of the initial sorbed boron concentration on the leaching behaviour.

Figure 11 and Figure 12 show boron concentration at five depths in the soil profile, i.e. 10, 30, 60, 100, and 200 cm (based on profile NAP7). When $\alpha$ is $10^{-4}$ times smaller than its measured value (Figure 11), boron concentrations are released from the sorption sites into the pore water which explains the initial increase in concentration, especially at the 100 and 200 cm depth. The latter two
depths have initially the largest adsorbed concentration, hence the release into the pore water is largest. Their concentrations drop towards the end of the warming up period. At the shallowest two depths, concentration remain rather constant during the warming up period. Their initial adsorbed concentration is the lowest, with the release into the pore water being much more modest compared to the deeper depths. Once the irrigation starts, boron concentrations increase throughout the entire profile, as boron is added to profile with each irrigation event. The largest increase is observed at the shallowest depths (10 and 30 cm), up to a factor 3. At the largest depths the increase is less than a factor 2. These calculated concentrations are somewhat lower than the measured pore water concentrations, as discussed in section 4.7.3.2.2 and Figure 3: average concentrations were 5.6 mg/L for the 0-10 cm interval, 1.3 mg/L for the 10-30 cm interval, and 2.07 mg/L for the 30-60 mg/L interval. While there is considerable spatial variability in the chemical parameters and in the boron concentrations, something that was not entirely captured by the sampling and analysis protocol, other reasons contribute to the underestimation of the boron concentrations. These include insufficient data to derive adequate parameters that describe the complex processes of boron adsorption-desorption, including lack of parameters that describe the longer-term exchange processes between solid and liquid phase. The current simulations are therefore preliminary results that need further corroboration, especially to get more representative sorption parameters.

In the second set of calculations the exchange parameter $\alpha$ was further reduced to $10^{-5}$ times smaller than its measured value (Figure 12). The boron behaviour is similar to that of the previous simulations, in that the warming period produces now produces nearly constant values until start of the irrigation. As soon as the irrigation commences, boron concentration increases, up to about a factor 4 for the shallowest depths. The highest concentrations for the shallowest depths are again similar to those of the previous case, around 0.4-0.5 mg/L. In other words, while the warming up period is sensitive to the $\alpha$ value, the irrigation period is much less sensitive with boron mainly determined by the amount added and the sorption parameters of the Langmuir isotherm.

As part of the sensitivity analysis we tested the effect of initial sorbed boron concentration on boron leaching. Three scenarios are considered: minimum, mean, and maximum values of adsorbed boron concentration in hard red brown soil as reported in Table 4. Simulations are carried out with the $\alpha$ value set to $10^{-5}$ x its base value; this small values was shown to give the most realistic boron behaviour in the warming up period (Figure 12). Simulated boron concentrations at four depths (30, 60, 100, and 200 cm) for the 80-year simulation period are displayed in Figure 13. While the effect of variability in initial sorbed boron is clearly noticeable in the pore-water concentrations, the overall variability in simulated concentrations is rather small. At the shallow depth (30 cm) concentrations range from 0.3 to 0.5 mg/L B, whereas at 200 cm depth the concentrations range from 0.4 – 0.5 mg/L B. Note that the slightly larger variation at the shallow depth is influenced by the temporal variability in the water flux; deeper in the soil profile such variations typically become smaller (e.g. Mallants et al., 2017). Overall the boron concentration as a result of irrigation increases by about a factor of four compared to the warming up period (where small amounts of boron were added from rainfall, i.e. 0.05 mg/L). Note that through irrigation 0.33 mg/L B is added to the soil profile. This accounts for the total increase in B concentration, i.e. from about 0.1 mg/L to about 4.5 mg/L.

The variability in boron concentration in the irrigation water was also tested. Three scenarios are considered, with the following boron concentrations: 0.2 mg/L (minimum), 0.33 mg/L (mean), and 0.53 mg/L B (maximum), as per the statistical parameters from Table 2. As expected, simulated boron concentrations in the soil profile react in a linear way to the increased concentration in the irrigation water. For example, the maximum B concentration (0.53) in irrigation water is about 2.5 times larger than the minimum concentration (0.2 mg/L). As a result, the boron concentration in the soil for the former conditions is also about 2.5 times larger than for the latter, i.e. about 0.7 mg/L versus 0.3 mg/L (Figure 14). This is true for all depths. Interestingly, the variation in B concentration
as a result of variation in irrigation water quality is larger than the variation in B concentration due to variability in adsorbed boron. This illustrates that one of the factors to manage B in soil is through managing the B concentrations in the irrigation water.

In developing sorption models for boron different Langmuir isotherm parameters were derived that resulted in two types of sorption profiles, i.e. Type-1 and Type-2 (Table 7). The sensitivity of boron leaching towards these sorption models was tested by running two scenarios, one with the Type-1 and the other with the Type-2 data. As can be seen in Figure 15, boron leaching is not sensitive to the variation in these parameters (at least not for the variability considered here). Indeed, a nearly identical boron behaviour is observed for both scenarios at all soil depths.

By testing the sensitivity of boron leaching towards several key sources of variability (and thus uncertainty), we demonstrated that the leaching model is least sensitive to the natural soil variability (sorbed boron concentration and sorption models) and most sensitive to the variation in boron concentration in the irrigation water. Considering the mean boron concentration in irrigation water, simulations showed that boron in soil would increase by about a factor of four as a result of long-term irrigation. As the B concentration time series show, a quasi-steady state condition is achieved across all depths illustrating that there does not seem to be a long-term accumulation of B in the soil profile. Over time, an equilibrium is established between the boron added and that leaving the soil profile by drainage.

Figure 11 Boron concentration in pore water of hard red brown soils (Type-2 sorption parameters, Table 7). Two-site non-equilibrium model (Type-2) based on data in Table 11 and Table 10. Initial concentration Type-2 from Table 11. The $\alpha$ value was set to $10^{-4} \times$ its base value.
As can be expected, the final steady-state boron concentration in the soil pore water will also depend on the initial boron concentration prior to adding boron via the irrigation water. As the initial boron concentration (i.e., prior to adding boron containing irrigation water) in the pore water and on the solid phase increases, the effect on the long-term boron concentration will diminish. This means that soils with an already high boron concentration will be at a lesser risk relative to soils with a much lower boron concentration. For instance, initial pore water boron concentrations in the range 0.2-0.3 mg/L may see an increase of a factor 2 at most. For initial boron concentrations of about 1 mg/L, the increase is not more than 40% (results not shown).
Figure 13 Sensitivity analysis of boron leaching in hard red brown soil (Type-2 sorption parameters, Table 7). Effects of using different initial sorbed B concentration (minimum, mean, maximum from Table 4). The $\alpha$ value was set to $10^{-5} \times$ its base value.
Figure 14 Sensitivity analysis of boron leaching in hard red brown soil (Type-2 sorption parameters, Table 7). Effects of using different B concentration in irrigation water. Mean = 0.33 mg/L B, Min = 0.2 mg/L B, Max = 0.53 mg/L B. The $\alpha$ value was set to $10^{-5} \times$ its base value.
Figure 15 Sensitivity analysis of boron leaching in hard red brown soil. Effects of using different Langmuir sorption isotherm parameters (Type-1 and Type-2 sorption parameters, Table 7). The α value was set to $10^{-5}$ × its base value.
4 Conclusion

Modelling long-term boron transport in soil is complicated by a number of factors, including rates of mineral dissolution, adsorption-desorption processes, including whether or not sorption is linear or non-linear, instantaneous or kinetically controlled. Based on best available data on boron in soils from the NAP region, the boron adsorption processes were derived by considering the following conceptual model:

- Boron adsorption/desorption is governed by a two-site model, where boron is distributed across equilibrium sites ($S_1$: sorption is instantaneous) and kinetically controlled sites ($S_2$: sorption is time-dependent). Best-estimate parameter values were derived for the fraction $f$ of equilibrium sites $S_1$ and the kinetic mass exchange parameter $\alpha$ for $S_2$ sites. Parameter $f$ was found to vary between 0.32 (most sites are kinetically controlled, i.e. $S_2 > S_1$) and 0.74 (most sites display instantaneous sorption, i.e. $S_1 > S_2$). The $\alpha$ parameter had values between 0.1 and 0.31 day$^{-1}$. However, because $\alpha$ was derived from short-term desorption tests (96 hrs), the values are thought not to be representative for calculating long-term boron behaviour in soil. For this reason this parameter was arbitrarily decreased until realistic boron behaviour was simulated. Future work should address this uncertainty by developing longer-term desorption tests that produce kinetic parameters for long-term simulations.

- Boron isotherms were derived as either linear (the $K_d$ model) or non-linear (Langmuir or Freundlich model). The best estimate parameters from the Langmuir model were used in the simulations, as these provide greatest flexibility in describing the complex sorption process.

Several modelling scenarios were undertaken and mainly served as a sensitivity analysis, given the uncertainty around key parameters such as the $\alpha$ parameter. By testing the sensitivity of boron leaching towards several key sources of variability (and thus uncertainty), we demonstrated that the leaching model is least sensitive to the natural soil variability (sorbed boron concentration and sorption models) and most sensitive to the variation in boron concentration in the irrigation water. Considering the mean boron concentration in irrigation water, simulations showed that boron in soil could increase by about a factor of four as a result of long-term irrigation. As the B concentration time series show, a quasi-steady state condition is achieved across all depths illustrating that there does not seem to be a long-term accumulation of B in the soil profile. Over time, an equilibrium is established between the boron added and that leaving the soil profile by drainage.

Interestingly, the variation in B concentration as a result of variation in irrigation water quality is larger than the variation in B concentration due to variability in adsorbed boron. This illustrates that one of the key factors to manage B in soil is through managing the B concentrations in the irrigation water. The current simulations are preliminary results that need further corroboration, especially to get more representative sorption parameters.

Importantly, as the initial boron concentration (i.e., prior to adding boron containing irrigation water) in the pore water and on the solid phase increases, the effect on the long-term boron concentration will diminish. This means that soils with an already high boron concentration will be at a lesser risk relative to soils with a much lower boron concentration.
5 References


Appendix 11 Boron risks associated with recycled water irrigation


Appendix 12: Climate extremes and their impacts on crop growth and irrigation requirement

Contents

Executive summary .............................................................................................................................. 411
1. Introduction ...................................................................................................................................... 413
2. Literature review .............................................................................................................................. 414
3. Northern Adelaide Plains (NAP) study area .................................................................................. 417
4. Selection of climate indices ............................................................................................................. 418
5. Application of climate indices to the NAP region .......................................................................... 419
5.1. Daily temperature, number of hot days and hot spells ................................................................. 419
5.2. Number of dry days and dry spells ................................................................................................. 421
5.3. Number of cold days ..................................................................................................................... 423
6. Crop specific impacts and mitigations ............................................................................................. 425
6.1. Horticultural crops (potatoes, carrots, onions) ............................................................................ 425
6.2. Perennial horticulture (vines, almonds, pistachios) ................................................................. 427
6.3. Broad acre crops (lucerne, pasture) ............................................................................................. 433
7. Irrigation requirement estimated with FAO-56 ............................................................................. 435
8. Climate change effect on plant water stress .................................................................................. 442
9. Conclusions ....................................................................................................................................... 448
References ............................................................................................................................................ 450
Executive summary

Agricultural industries such as viticulture, perennial and annual horticultures can be impacted by climate extremes. In this study, the effect of climate extremes on crop growth and irrigation requirement in the Northern Adelaide Plains (NAP), South Australia (SA), was investigated. The climate indices for historic (1985-2017) and future (2018-2050) climate were analyzed to determine the frequency of extreme climate events, and its impact on the growth of horticultural crops (potatoes, carrots, and onions), fruit trees (vines, almonds, and pistachios), and broad acre crops (lucerne and pasture).

Using greater daily temperature indices and frequency of hot days relative to historic climate, results showed that the NAP region will be subjected to significant warming in the near future. There will be also a higher frequency for dry days (with rainfall less than 1 mm) and dry spells compared with historic data. The NAP region, following global climate change predictions, will be subjected to milder winters with smaller frequencies of frost and chill days.

The following consequences are anticipated for the crops in the NAP region, as the result of these climatic shifts:

- Potatoes will likely see a yield decrease and a higher risk to being invaded by pests;
- The growth and yield in carrots will be stimulated due to increased frequency of hot and dry days. However, extreme heat events may reduce the quality of carrots and mid-season drought stress can depress the yield in carrots.
- Warmer climate may reduce the duration of crop growth, yield, and seed production for onions; whereas, the low rainfall condition can reduce the risk of infection by pests (i.e., leaf blight).
- The projected drought and extremely hot weather in the NAP region can negatively impact vines, which may result in poor budburst, leaf loss, bunch damage, and consequently low yield and production or even crop loss. Warmer summers may lead to adopting warmer climate grape varieties.
- For almonds and pistachios, it is expected that their yield and production may be impacted by drought in the NAP region if not properly managed. While the current NAP climate accommodates chill requirements for these fruits, the projected climate shows there would
be some years that this requirement cannot be met at all, which presents challenges to growers with changes in management practices and perhaps even new cultivars needed to accommodate new growing conditions.

- As the NAP region will be subjected to an increased number of hot days, a decline in pasture production is anticipated.

In order to understand the effect of weather extremes on irrigation practices, the irrigation requirement for abovementioned crops was calculated using the FAO-56 dual crop coefficient method (this report, Annex 7). The irrigation requirements for these crops were compared under historic and future climate scenarios to provide insights of future climate impact. Results showed that crops will require a higher amount of annual irrigation under future climate, depending on the soil textures and crop stress tolerance. Regardless of the crop type, sand over clay soils require the highest irrigation (714 – 956 mm) while deep uniform to gradational soils need the lowest irrigation (643 – 910 mm). It was concluded that annual horticultural crops could face more irrigation related risks in the future climate compared to deep rooted perennial horticultural crops.

The results of water balance simulation showed that under future climate, pastures on deep uniform to gradational soils will experience 243 extra water stress days (over a 32 year period) compared to historic climate, or 7.6 days per year. Over the 32 years period, hard red brown soils will experience 105 extra water stress days or 3.3 per year under future climate. This implies that pasture’s yield and production will be reduced if not irrigated.
1. Introduction

The observed global climate trends show an increase in air temperature, decrease in oceanic waters’ pH, and sea level rise over the past three decades, providing strong evidence of a global climate change (Jones and Moberg, 2003; Church and White, 2006, 2011; Steffen, 2006; Fisher, 2007; Vose et al., 2012; IPCC, 2013; Marcott et al. 2013; Allen et al., 2018). The consequences associated with climate change include, but are not limited to, hotter and drier summers, milder winters, reduced rainfall events, water resources’ scarcity, increased risk of flooding and coastal erosion due to sea level rise, risk to marine calcifying organisms such as oyster, mussel, coral and plankton varieties as a result of decrease in oceanic waters’ pH, decrease in food security due to impacts to agricultural industries, as well as increase in frequency of extreme weather events such as strong winds, lightning, hail storms, drought and heat waves (Fisher, 2007; Australian Bureau of Meteorology, 2008, 2009a, 2009b, 2010, 2011; Wheeler and von Braun, 2013, Thomas et al., 2016).

Agricultural industries such as viticulture, perennial and annual horticultures, and annual cropping are key industries which can be impacted by climate extremes (White et al., 2006; Thomas et al., 2016; Ziska et al., 2016). Extreme climate conditions such as drought, heatwaves, extreme heat and cold (or frost), expose a great risk to wine grape production (White et al., 2006; Shellie and Glenn, 2008; Wittwer, 2008; Australian Bureau of Meteorology, 2008, 2009a, 2009b, 2011; Jones et al., 2010; Crimp et al., 2012, 2015; Araujo et al., 2016), and crop plants’ growth and yield (Garrett et al., 2006; McMichael et al., 2007; Lobell and Field, 2007; Thornton, 2012; Rehman et al., 2015; Zhu and Troy, 2018). To mitigate the vulnerability to climate change, it is required to understand the impacts of near-term climate change on the supply of key food commodities through analyzing past climate trends (Lobell et al., 2011). Through the proper planning, management, and irrigation practices, negative impacts of climate extremes on the crop growth and production can be minimized efficiently. Moreover, identifying which particular crops and regions have been most impacted by extreme climate conditions would assist us to measure and analyze their impacts and adapt accordingly (Lobell et al., 2011). The expanded rainwater harvesting, water storage and conservation, efficient irrigation, adjusting planting dates and crop variety, and relocating crops (IPCC, 2013; Haverkort and Verhagen, 2008) are among the adaptation techniques which can be used to minimize these negative impacts.
This research was aimed to investigate the effect of climate extremes on crop growth and irrigation requirement in the Northern Adelaide Plains (NAP), South Australia (SA). The climate indices for historic (1985-2017) and future (2018-2050) climate were analyzed to determine the frequency of extreme climate events and its impact on the growth of horticultural crops (potato, carrot, and onion), fruit trees (vines, almonds, and pistachio), and broad acre crops (lucerne and pasture). First, the irrigation requirement was calculated by means of the FAO-56 dual crop coefficient method for the abovementioned crops to provide insights of future climate on irrigation requirement and impacts of weather extremes on these crops. Next, the soil water availability was calculated with the mechanistic soil water balance simulator HYDRUS-1D for the reference crop pasture. For the calculation and comparison of climate statistics for historic and future climate, two time series of identical length (i.e. 32 years) were used.

2. Literature review

Due to the importance of agricultural crops, a broad range of studies has been conducted to investigate the impact of climate change on crop growth and yield. In general, these studies have emphasized on negative impacts of weather extremes (e.g., high temperature, low rainfall) on the production of crops, especially during the growing period (Reilly et al., 2002; Fischer et al., 2005; Lobell and Field, 2007; Thornton, 2012; Haile et al., 2017). However, climate change can lead to changes not only in crop yields (Adams et al. 1990; Reilly et al. 2002; Peng et al., 2004; Deschenes and Greenstone, 2007) but also in crop yield variability (Chen et al., 2004; McCarl et al., 2008). The negative consequences associated with high temperatures include reduced critical growth periods of crops (Haile et al., 2017), more frequent crop diseases (Coakley et al., 1999; Garrett et al., 2015; Haile et al., 2017), damage to reproductive tissues of plants, and increased pollen sterility (Roberts and Schlenker, 2009; Thornton, 2012). Drought (low rainfall) or excessive rainfall not only affect the crop production, but also yield and acreage effects (McCarl et al., 2008; Haile et al., 2017).

Previous studies (Rosenzweig and Parry, 1994; Lobell and Field, 2007, 2011; Haile et al., 2017) have been mostly concentrated on the impact of the climate change on the world’s key staple crops
Appendix 12 Climate extremes and their impact on crop growth | 415

(e.g., wheat, rice, maize, soybeans, barley and sorghum), due to their importance as a major source of food around the world. The first global assessment of the potential climate change impacts on staple crops was conducted by Rosenzweig and Parry (1994). Using numerical models for wheat, rice, maize, and soybeans, they anticipated positive yield changes for countries that are positioned in middle and high latitudes and negative yield changes for countries that are located in low latitudes as a result of climate change. Lobell and Field (2007) used empirical/statistical models to understand the relationship between climate and crop yield at a global scale, and concluded that recent climate trends have had a negative impact on the global production of major crops such as wheat, rice, maize, soybean, barley, and sorghum. A regression model was used by Lobell and Field (2011) to understand the effect of climate on yield outcomes for staple crops. They concluded that, in most situations, crop yields are more sensitive to temperature extremes than precipitation extremes. Their study, in agreement with the previous study by Rosenzweig and Parry (1994), showed that a 1°C rise tended to decrease yields up to 10% except in high latitude countries, where in particular rice gains from warming. Increases in precipitation, on the other hand, increases yields for nearly all crops and countries (Rosenzweig and Parry, 1994; Lobell and Field, 2011). More recently, Haile et al. (2017) developed an empirical model to investigate the impact of climate change and weather extremes on crop production, and predicted that climate change could reduce the production of staple crops by 9% in the 2030s and by 23% in the 2050s, globally. Their study found that increasing mean growing season temperature may not pose a major problem to crop production. Unlike mean temperature, increasing temperatures at the two extremes (e.g., higher minimum temperature for rice and higher maximum temperature for maize) does negatively impact the crop production. A similar situation exists for wheat and soybeans, where a higher average temperature becomes problematic after a certain critical level. A drawback of all these models is that they fail to simulate future yield responses when cropping areas shift, as evidenced by the recent expansion of soybean area in Brazil (FAO, 2006), or when the range of future temperatures exceeds those for which the models were calibrated (Lobell and Field, 2007).

In comparison, fewer studies have been conducted to investigate the impact of climate extremes on the growth of horticultural crops (potato, carrot, and onion), fruit trees (vines, almonds, and pistachio), and broad acre crops (lucerne and pasture). Reidsma et al. (2015), using an integrated assessment (Rotmans and Van Asselt, 1998; Van Ittersum et al., 2008) and farming systems
Appendix 12 Climate extremes and their impact on crop growth | 416

analysis (Janssen and van Ittersum, 2007), showed that while impacts of climate change on potato yields at a farm level (Dutch province Flevoland) are positive, the increased frequency of extreme events (e.g., heat waves) can turn positive impacts into negative ones.

However, adaptation techniques such as drip irrigation and planting in wider ridges can reduce negative impacts. Furthermore, recent studies (e.g., Pulatov et al., 2015) suggested that climate change may shift the planting/harvesting time for potato, depending on the geographic location of the country. For example, planting of potato at an earlier date could be possible in the warm climate regions. This is desirable as it can reduce the risk of summer drought (Supit et al., 2012), attacks by Colorado potato beetle (Jönsson et al., 2013; Pulatov et al., 2014), and infestation by late blight (Haverkort and Verhagen, 2008; Wiik, 2014). However, in cold climate regions such as north Europe, earlier potato harvesting may expose a high risk of frost damage (Haverkort, 1990; Pulatov et al., 2015).

For determinate crops such as onions, warming can reduce the duration of crop growth and hence yield (Idso, 1990; Wheeler et al., 1994, 1996; Harrison et al., 1995; Daymond et al., 1997; Wurr et al., 1998), while it may stimulate growth and yield in indeterminate crops such as carrots. For onions, warmer future climate scenarios may lead to yield decrease and cooler future climate scenarios may result in yield increase (Harrison et al., 1995; Maracchi et al., 2005). The situation is reverse for carrots that show an increased yield in warmer climate conditions and a decreased yield in cooler conditions (Wurr et al., 1998; Lal et al., 2018).

Extreme climate conditions can mean significant concerns for vines, if management strategies are not employed. Previous studies (Hayman et al., 2012) showed that vines flowering during extreme events are poor in fruit production and consequently low in yields. Serious negative consequences have been reported from vineyards in South Australia and Victoria as a result of heatwaves, including poor budburst, leaf loss, bunch damage, and even crop losses (The Australian Wine Research Institute, 2018). A simulation study by Araujo et al. (2016) suggested that grape yields are more sensitive to spring and summer droughts in the absence of irrigation. There are also additional studies showing that warmer and drier weather conditions may shift the produced wine grapes to warmer climate grape varieties (White et al., 2006; Thomas et al., 2016). Indeed,
heatwaves can result in sunburnt fruit along with rapid sugar level increases. Higher than expected sugar levels will lead to higher than desirable alcohol levels which can impact on yeast’s ability to complete fermentation, therefore affect wine style and potentially lead to loss of wine quality. Also, increased temperatures and periods of hot dry weather will lead to faster ripening and shifting forward harvest dates (The Australian Wine Research Institute, 2015).

Climate change is likely to also affect chilling and plant dormancy, which are vital for fruit trees including almonds and pistachios (Schwartz, 1999; Baldocchi and Wong, 2008; Luedeling and Brown, 2011; Luedeling et al., 2011; Luedeling, 2012). Previous research has shown dramatic losses in number of chilling hours for warm growing regions such as South Australia (Luedeling, 2012).

Soil water availability, water flux, root water uptake, and evapotranspiration (evaporation and transpiration) are intimately coupled processes influencing crop growth and yield. Previous studies have generally not taken into account the effect of climate extremes on these processes. Therefore, the present work attempts to fill this gap in knowledge and will first assess the impact of climate change on irrigation requirement for horticultural crops (potato, carrot, and onion), fruit trees (vines, almonds, and pistachio), and broad acre crops (lucerne and pasture) in the NAP region. Next, the soil water availability is compared between historic and future climate and conclusions are drawn about consequences for crop growth.

3. Northern Adelaide Plains (NAP) study area

The Northern Adelaide Plains (NAP) represents a highly diverse region with active viticulture, horticulture, annual cropping, livestock, and dairy industries (Thomas et al., 2016). This region benefits from a variety of fertile soils, including well-drained sandy soils and heavier clay soils; high quality groundwater is supplied by confined sand and limestone aquifers (Pitt et al., 2013). In addition, the region has been recently (since 1999) equipped with the Bolivar wastewater treatment plant which provides tertiary-treated wastewater via a pipeline network as a supplementary source of water for crop irrigation (Pitt et al., 2013; The Goyder Institute for Water Research, 2016; The Australian Water Association, 2019). These features along with
Mediterranean-type climate (with hot, dry summers and mild, wet winters) make the NAP a suitable region for cultivation of horticultural crops, fruits, and broad acre crops. However, the NAP region, similar to many regions in the world, is vulnerable to climate change and proper management and irrigation practices are required to mitigate the negative impacts of climate change on crop production.

4. Selection of climate indices

A total of ten climate indices were considered in this study and include i) mean of daily average temperature, ii) mean of daily maximum temperature, iii) mean of daily minimum temperature, iv) mean difference between the daily maximum and minimum temperature, v) the number of days when daily maximum temperature exceeds 35°C (extremely hot days), vi) the number of hot spell1s, vii) the number of days when daily precipitation drops below 1 mm (dry days), viii) the number of dry spells2, ix) the number of days when daily minimum temperature goes below 0°C (extremely cold or frost days), and x) the number of days when daily minimum temperature in winter is below 7.2°C (chilling days). These indices were calculated using climate parameters such as temperature, precipitation, and humidity for historic (1985-2017) and future (2018-2050) climate data. Note that to have the same number of data points in historic and future time series, both time series were limited to 32 years each.

The climate data used is based on downscaled climate change projections for South Australia (Charles and Fu, 2014; The Goyder Institute for Water Research, 2019a), generated from a subset of global climate models (GCMs). These models were chosen because they were found to perform better at representing climate drivers that are particularly influential on rainfall in South Australia, such as the El Nino Southern Oscillation (ENSO) and Indian Ocean Dipole. The projected climate data are the local scale climate projections, which includes rainfall, temperature, solar radiation, vapor pressure deficit and evapotranspiration data for South Australia (The Goyder Institute for Water Research, 2019b).

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1 Three to five consecutive days above 35 °C; six or more days above 35 °C are counted as two hot spells.
2 Three or more consecutive days with less than 1 mm of rain
5. Application of climate indices to the NAP region

In this section, climate indices were analyzed to determine the frequency of extreme climate events and their spells for historic and future climate in the Northern Adelaide Plains (NAP), SA.

5.1. Daily temperature, number of hot days and hot spells

In order to compare the historic and future temperature regimes, daily temperature indices including annual mean of daily average temperature, annual mean of daily maximum temperature, annual mean of daily minimum temperature, and annual mean difference between the daily maximum and minimum temperature (Table 1) were calculated for historic and future climate. Future temperature indices are all greater than historic ones, providing an evidence that the NAP region will be subjected to warming in the near future.

Previous studies (e.g., IPCC, 2018) have already warned about negative consequences, potential impacts and associated risks of warming, which include increases in mean temperature in most land and ocean regions, hot extremes in most inhabited regions, heavy precipitation in several regions, and the probability of drought and precipitation deficits in some regions (IPCC, 2018). It is anticipated that the NAP region will experience some of these negative consequences, if not all, as it will be subjected to warming in the future.

Table 1 Daily temperature indices for historic (1985-2017) and future (2018-2050) climate. a, b: mean values are significantly different at the 95% level based on t-test.

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Figure 1 shows the number of days with maximum daily temperature at or above 35°C (hot day) for historic and future climate. For only one statistic (95th percentile) does the historic climate produce larger number of days compared to the future climate. This is because in the year 2004 there was a record number of hot days (95th percentile = 29) that exceeded those of any future year.
(95th percentile = 21). In comparison with historic climate, future climate showed a significantly greater number of days (449 versus 379) with maximum temperature at or above 35°C (Table 1).

In addition to single days above the threshold, hot spells (3 or more consecutive days > 35°C) were also calculated. Although spells at or above 35°C for future data (21 spells) is slightly less than that of the historic data (24 spells), there is now also the possibility of hot spells occurring at or above 40°C. The cumulative distribution function of the number of days in any year when temperature exceeds 35°C (Figure 2) shows that there is approximately an equal shift towards a greater number of days > 35°C across the full distribution function (for nearly all percentiles). As mentioned above, an exception exists at the 95th and higher percentile because of an exceptionally hot 2004.

Figure 1 Number of days with a maximum temperature above 35°C for historic (1985-2017) and future (2018-2050) climate.
Table 1 Number of hot days and hot spells for historic (1985-2017) and future (2018-2050) climate.

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<td>Days at or above 35°C</td>
<td>379</td>
<td>449</td>
<td>24</td>
<td>21</td>
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<td>Spells at or above 35°C</td>
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<td>Days at or above 40°C</td>
<td>46</td>
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<td>Spells at or above 40°C</td>
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Figure 2 Cumulative distribution function for number of days in a year when temperature exceeds 35 °C.

5.2. Number of dry days and dry spells

A dry day is defined when daily rainfall is less than 1 mm. To determine the frequency of dry days, the cumulative distribution function (CDF) was plotted (Figure 3) for both historic and future climate data. Figure 3 indicates a greater occurrence of dry days in the future compared with the historic data, with generally an equal shift towards more dry days across all percentiles. For example, on average (50th percentile) the number of dry days increases from 290 to 296. This is an increase by about 2.1%. Additionally, spells of three or more dry days (dry spells) with rainfall
less than 1 mm for historic and future climate were also calculated (Refer to Table 2 and 4). On average (50th percentile) the number of dry spells increases from 81 to 86. This is an increase by about 6.2%.

![Graph showing cumulative distribution function for number of days (per year) with a rainfall less than 1 mm for historic (1985-2017) and future (2018-2050) climate data.]

*Figure 3 Cumulative distribution function for number of days (per year) with a rainfall less than 1 mm for historic (1985-2017) and future (2018-2050) climate data.*

*Table 2 Total number of dry days and dry spells for historic (1985-2017) and future (2018-2050) climate.*

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<tr>
<td>Dry days</td>
<td>9603</td>
<td>9805</td>
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<td>2808</td>
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5.3. Number of cold days

A frost day is defined when daily minimum temperature goes below 0°C. Based on the CDF of days with temperature less than 0°C for both historic and future climate data, Figure 5 shows that the number of frost days in the future will be less than the historic data, confirming that the NAP region, following a global climate change, will be subjected to milder winters. It should be noted, however, that the dryer climate could increase the severity of frost despite their reduced frequency. This could have serious consequences for crops whose phenology has been advanced or compressed due to the milder winters/springs. As an example, changes to almond phenological timing could impact the synchronization of pollinator varieties with main commercial varieties.

Figure 4 Cumulative distribution function for number of spells (3 days or more with rain < 1 mm) per year with for historic (1985-2017) and future (2018-2050) climate data.
Fruit trees such as almonds and pistachios require a minimum period of cold winter weather after which a fruit-bearing tree will blossom. The chilling requirement can be calculated as chilling hours (or chilling days), which is the sum of total amount of time in a winter spent at certain temperatures (Lockwood and Coston, 2005; Texas A&M University Agrilife Research & Extension, 2019). The adequate amount of winter chilling results in homogeneous and simultaneous flowering (Luedeling et al., 2009a; PGAI, 2019). In SA, almonds (Pitt et al., 2013) require a winter chilling period of 400–900 hours (≈ 16.5–37.5 days) and pistachios require winter a chilling period of 600–1050 hours (≈ 25–43.75 days) below 7.2 °C (Küden et al., 1994) to initiate flowering. Figure 6 shows that the annual number of chilling days in the future will decrease on average by 4 days (or about 7.7 %). Table 3 shows that the total number of frost days is only half that of the future climate, while the total number of chilling days decreased by almost 150 days.

Figure 5 Cumulative distribution function for annual number of frost days (temperature less than 0°C) for historic (1985-2017) and future (2018-2050) climate data.
6. Crop specific impacts and mitigations

6.1. Horticultural crops (potatoes, carrots, onions)

Potato is considered as the fourth most important food crop after rice, wheat and maize (Pulatov et al., 2015). In SA, approximately 11,900 ha of land has been allocated for potato growth providing an average annual production of over 300,000 tonnes (ABS, 2012; Pitt et al., 2013). The NAP region contributes 14% to SA potato industry (Pitt et al., 2013), which highlights its critical role in supplying this crop.
Potatoes can be grown in many months of the year, provided it has 60-90 days of frost-free conditions to be successfully harvested. The best planting times is March-April, as the soil is warm, growth is rapid and there are generally less pests. The NAP region has two main harvest periods, i.e. from June to August and from November to February.

With an increased frequency of hot and dry days (and dry spells) in the future (Figure 1 and Figure 2), negative impacts are anticipated for potato including a yield decrease and a higher risk to being invaded by pests such as Colorado potato beetle and late blight which can reduce the quality of produced potatoes. Research (Haverkort & Verhagen, 2008) has shown that potatoes’ growth slows down significantly when the daily average temperature falls below 5 °C or rises above 21 °C, and their photosynthetic capacity halts completely at temperatures below 2 °C and above 35 °C. Projected temperature patterns (Figure 1) in the NAP region show that potatoes may suffer from lack of photosynthetic capacity (and hence low yield) due to frequent hot days with temperatures above 35 °C, which mainly occurs throughout January and February. In the NAP region, the potatoes that are harvested from November to February, are most at risk and may require earlier harvest to avoid the effects of hot days.

In order to mitigate the negative impacts of the shortage in rainfall, more efficient irrigation techniques, such as drip irrigation and planting in wider ridges can be used. To reduce the negative impacts associated with high temperatures, potatoes can be planted/harvested at an earlier date, as suggested by recent studies (Haverkort and Verhagen, 2008; Supit et al., 2012; Wiik, 2014; Jönsson et al., 2013; Pulatov et al., 2014, 2015).

More than 30,000 tonnes of carrots are produced in NAP region yearly, which makes the carrot one of the most economically important crops in the region (ABS, 2012; Pitt et al., 2013). Packaging and processing facilities located in the Adelaide Plains process nearly half of the SA carrots (Pitt et al., 2013). Indeterminate crops such as carrots are less sensitive to heat stress compared to determinate crops such as onions. From the projected future temperature (Figure 1), it is expected that warmer temperature may stimulate the growth and yield in carrots (Harrison et al., 1995; Wheeler et al., 1996; Wurr et al., 1998; Olesen and Bindi, 2002; Maracchi et al., 2005).
However, extreme heat events may reduce the quality of carrots as a result of acrid flavor development (Simon, 1985) and textural deterioration (Pitt et al., 2013). Warmer climate may expose a further disadvantage for carrots, as it then becomes their tendency to invest energy into foliage rather than tuber growth (Wurr et al., 1998). Carrots do not need irrigation during early growth, and dry soil conditions during this stage even increases the yield (Dragland, 1978). However, mid-season drought stress can depress the yield in carrots. Dry soil conditions during early growth may also influence their chemical composition (Dragland, 1978; Sørensen et al., 1997) and increase the risk of infections by common scab (Schoneveld, 1991). Advanced management and irrigation techniques are required to handle negative consequences of heat and drought stresses for vegetable crops such as carrots, grown in broad acre areas. Use of center pivot (instead of fixed overhead sprinkler) is an example of an efficient method, which has been recently used for irrigation of carrots. This method successfully reduced the disease buildup, and enhanced the end-product quality (Government of South Australia, 2006).

In the NAP region, more than 13,000 tonnes of onions are produced annually which are planted on 206 ha (ABS, 2012). Onion is considered a cool season vegetable crop, which can be more adversely affected by temperature extremes than some warm season crops (McKoewn et al., 2004; Hazara and Som, 2009; Lal et al., 2018). It is anticipated that warmer climate reduces the duration of crop growth, yield, and seed production for onions (Idso, 1990; Wheeler et al., 1994, 1996; Harrison et al., 1995; Daymond et al., 1997; Wurr et al., 1998; Maracchi et al., 2005; Lal et al., 2018). It should be noted that low rainfall (or dry) condition can reduce the risk of infection by leaf blight (Msuya et al., 2005), which is a common disease in onions. To alleviate the negative impacts of the climate extremes on onions, it might be a useful strategy to cultivate a variety of onions that is more tolerant to weather extremes. For example, the dry onion (Allium cepa L.) might be a good choice as it can be grown in a wide range of climatic conditions from relatively hot and dry areas to fairly cool and humid zones (Tindall, 1983; Messiaen, 1992).

### 6.2. Perennial horticulture (vines, almonds, pistachios)

In 2018, approximately 3000 tonnes of wine grapes were produced in the NAP region (Wine Australia, 2018) planted on about 600 ha (ABS, 2012). A large percentage of the produced wine grapes are red variety comprising largely Shiraz (Wine Australia, 2018). The projected extreme
climate conditions such as drought and extremely hot weather in the NAP region (Figure 1 and Figure 2) can negatively impact vines. This may include poor budburst, leaf loss, bunch damage, and consequently low yield and production or even crop loss (Hayman et al., 2012; Araujo et al., 2016; The Australian Wine Research Institute, 2018). In addition, temperatures greater than 35°C can reduce the amount of skin anthocyanin, the principal component in grapes responsible for wine color (Spayd et al., 2002; Shellie and Glenn, 2008). Warmer summers may lead to adopting warmer climate grape varieties (e.g. Marsanne or Viognier from Southern France, Tempranillo from Spain or Fiano from Southern Italy) to avoid poorer quality wines (White et al., 2006; Thomas et al., 2016), as they shift the grape ripening from milder fall and late summer to hotter mid-summer.

Efficient management and irrigation strategies should be employed to mitigate negative consequences that arise from extreme climate conditions. Alternative irrigation methods such as deficit irrigation can be used to manage vegetative and reproductive growth of wine grapes (and other perennial fruit crops), and enhance the product’s quality or to increase water use efficiency (Shellie and Glenn, 2008). The application of exogenous compounds such as kaolin particle film (Al₂Si₂O₅(OH)₄) to grapevines is suggested during abiotic stress periods, as it results in lower canopy temperatures and stomatal conductance, higher protection of photosystem structure and function in leaves exposed to excessive heat and solar radiation (Glenn et al., 2010; Dinis et al., 2016a, b, 2017; Conde et al., 2018). The other advantages of applying kaolin particle film is that it can promote total soluble solids and anthocyanin amount in grape berries, improve the molecular mechanisms involved in phenolics’ synthesis (Shellie and Glenn, 2008; Ou et al., 2010; Song et al., 2012; Shellie, 2015; Conde et al., 2016, 2018), enhance the sucrose synthesis and its transport capacity in leaves when the temperature exceeds 30 °C (Conde et al., 2018), and substantially improve the grape leaf metabolome that can tolerate the constraints of excessive summer stress (Conde et al., 2018) such as water deficit and high temperature with a better capacity.

Australia is considered the second major producer of the almonds (7%) in the world (ABA, 2017) after the United States (80%). Almonds are commercially cultivated in three states (Victoria, South Australia and New South Wales) of Australia (The Almond Board of Australia, 2012). They are best suited to areas with cool to cold winters and hot, dry summers, such as southern Australia. In the NAP region 724 ha of land has been used for almond cultivation (ABA, 2017), accounting for
2% of the Australian almond industry (Pitt et al., 2013). Although almond is considered as a drought resistant species (De Herralde et al., 2003; Rouhi et al., 2007) which is able to withstand frequent periods of low soil moisture accompanied by high evaporation rate and temperature during the growing season, its production and survival can be affected by the occurrence of cyclic and severe droughts (Ghrab et al., 2008). Under projected shortage of rainfall for 2018-2050, it is expected that yield and production of almonds could be impacted by drought in the NAP region if not properly managed.

Almonds require winter a chilling period of 400–900 hours (≈ 16.5–37.5 days) below 7.2 °C (Pitt et al., 2013) to initiate flowering. The analysis of historic and future climate data (Figure 6) shows that the number of chilling days is expected to decrease in the future. The current NAP climate still accommodates this chill requirement (Figure 7A). However, projections (Figure 7B) for 2018-2050 show that there would be some years (e.g., 2037) that chilling requirement cannot be met for the almonds and warmer winters will bring chill accumulation closer to the lower end of almond requirements. Previous research (Pitt et al., 2013) in the NAP region has shown that once flowering commenced, the warmer spring is likely to be advantageous for pollination and nut development, as well as reduced pressure from disease. One of the problems associated with increased frequency of extreme heat events (Figure 1) in summer is bud initiation during the post-harvest development phase, which may affect the following year’s crop (Pitt et al., 2013). Increasing irrigation is suggested to reduce the negative impacts of high summer temperatures.

The use of the number of chilling hours below 7.2 °C was found to overstate the loss of chill under warmer conditions, especially in warmer sites (Leudeling and Brown 2011). An alternative approach uses a dynamic model (Zhang and Taylor, 2011) to estimate chill portions; for almonds the minimum required chill portions depend on variety, ranging from 22 to 32 (Thomas et al., 2010). Thomas et al. (2010) calculated that for a +1°C temperature change, 10% less rainfall and 3% higher evapotranspiration, the mean chill portions on the NAP would be 40. This is still higher than the required value (32). These results are not materially different from those obtained based on the number of chilling hours below 7.2 °C: for all but one year is the chilling condition satisfied (Figure 7).
Figure 7 Number of chilling days for almonds. (A) During historic climate (1985-2017); (B) During future climate (2018-2050).
Pistachio trees were first planted in south-eastern Australia in the early 1980s (NPA, 2019; PGAI, 2019). In 1982, CSIRO introduced a new variety of pistachio tree, Sirora, which has a high yield and is well suited for Australian weather conditions. This variety includes special features such as an excellent flavour, bright green kernel colour, high percentage of wide splits (easy to open), as well as white shell (NPA, 2019). Pistachios are considered drought tolerant species (Elloumi et al., 2013); however, projected heat or drought stress (Figure 1 and Figure 3), may reduce the yield and lead to development of a hull cracking known as “early split” (Doster and Michailides, 1995; Hadavi, 2005; Cotty and Jaime-Garcia, 2007).

Similar to almonds, flowering and nut yield of pistachio trees is a function of chill accumulation (Elloumi et al., 2013). Previous studies have shown that warm winter temperatures with lack of chilling induce flower bud drop, underdevelopment of the pistil (Rodrigo, 2000; Rodrigo and Herrero, 2002), floral and leaf bud bursting delay (Arora et al., 2003), poor fruit set and low quality (George et al., 2002; Arora et al., 2003), irregular flowering and rosette formation (George et al., 2002), smaller terminal shoot extension and abnormal leaves (Breeuwer et al., 2008; Javanshah, 2010). Insufficient chill may also reduce pollination and crop yields (Rattigan and Hill, 1986; Gradziel et al., 2007; Baldocchi and Wong, 2008). Pistachios require winter a chilling period of 600–1050 hours (≈ 25–43.75 days) below 7.2 °C (Küden et al., 1994). While the current NAP climate (Figure 8A) hardly accommodates this chill requirement, projections for 2018-2050 show there would be some years that chilling requirement cannot be met for pistachios (Figure 8B). This may present challenges to growers with changes in management practices and perhaps even cultivars needed to accommodate new growing conditions (Luedeling and Brown, 2011; Luedeling et al., 2009b; Zhang and Taylor, 2011). It should be noted that development of a dynamic model (Fishman et al., 1987a, b; Zhang and Taylor, 2011) may assist farmers in determining the accurate amount of chill required for ‘Sirora’ pistachio grown in Australia, especially in the areas where the minimum winter temperature would be greater than 7.2 °C (Zhang and Taylor, 2011). The dynamic model, which incorporates non-stationary and time-inhomogeneous processes into chilling models, may better describe the bud break and yield responses of pistachio to chill in southeast Australia (Zhang and Taylor, 2011). This model (Fishman et al., 1987a, b; Zhang and Taylor, 2011) assumes that winter chill is accumulated in a two-step process. Initially, cold temperatures result in the formation of an intermediate chilling product which can be destroyed by high temperatures. Once a critical amount of this chilling product has accumulated, it converts to
a chill portion, which cannot be destroyed. A certain chill portion accumulation indicates fulfilment of chilling requirement.

![Figure 8 Number of chilling days for pistachios. (A) During historic climate (1985-2017); (B) During future climate (2018-2050).](image)

To overcome the problems of lack of winter chilling for pistachios, the application of hydrogen cyanamide is suggested, which has been demonstrated to advance and synchronize flowering of
male and female cultivars (Pontikis, 1989; Rahemi and Asghari, 2004). Previous research (Barone et al., 2005) has shown that applications of hydrogen cyanamide and urea improve the pistachio vegetative growth and maximum assimilation rate. In Australia, winter oil has been commonly used due to its major role in pistachio production during low-chill years (Zhang and Taylor, 2011). However, the time when the oil should be applied is a matter of debate since it can influence bloom date and yield in pistachios (Zhang and Taylor, 2011).

6.3. Broad acre crops (lucerne, pasture)

Lucerne (syn. alfalfa, Medicago sativa) is considered as the most important and widely grown fodder plant in the world since it provides an excellent source of protein and energy for livestock (AgriFutures Australia, 2019). As an herbaceous perennial legume, Lucerne is capable of fixing the atmospheric nitrogen and adding it to the organic nitrogen content of the soil which can remain in the soil for several years and contribute to the nutrition of subsequent crops or pastures (AgriFutures Australia, 2019). A deep tap root of up to 15m gives this plant several agronomic benefits, including survival in dry periods of summer due to its ability to use water at depth in the soil profile. The deep tap root has another benefit; that is, the plant can use water to alleviate rising water tables and improve soil structure (AgriFutures Australia, 2019). Lucerne is considered as a summer growing plant; however, plant breeding has extended the growing season, with many winter varieties available for cultivation (AgriFutures Australia, 2019). It is usually planted in autumn (irrigated Lucerne may also be planted in spring) when there is sufficient soil moisture and soil temperatures are warm enough (AgriFutures Australia, 2019).

Lucerne was introduced to Australia over 200 years ago. In Australia, it is currently cultivated across 3.2 million ha (Humphries, 2013) with more than 50 varieties available to Australian growers (AgriFutures Australia, 2019). Although this plant suits a wide range of soil types and environments from subtropical regions in Queensland to the cool climate of Tasmania, it grows best in Southern Australia (AgriFutures Australia, 2019). It can be grown as a short-term pasture in a cropping rotation to improve soil condition between annual crops and provide valuable forage, and to produce the fodder (high quality hay, silage or chaff) and seed.
The temperature ranges of 15–25°C for daytime and 10–20°C during the night, are considered as optimum temperatures for dry matter production (AgriFutures Australia, 2019). However, highly active winter cultivars can tolerate lower temperatures. As the NAP region will be subjected to an increased number of hot days, a decline in pasture production is anticipated for this region. This is in agreement with previous studies (Cullen et al. 2009; Moore and Ghahramani 2013; Thomas, 2016) which predicted that climate change will lead to a decline in pasture production in South Australia.

Winter frost can damage the new growth, but this may not influence the production of Lucerne due to its ability to reshoot from its crown throughout the season (AgriFutures Australia, 2019).

Lucerne is considered as a relatively drought tolerant plant (AgriFutures Australia, 2019), but its yield is a function of supplied water volume. It can survive with a minimum of 300–400 mm of annual rainfall in the environments ranging from Tasmania to northern New South Wales (AgriFutures Australia, 2019). A fodder production enterprise will have an irrigation water requirement of 7–13ML/ha, depending on location. Seed production enterprises operate successfully in south east Australia in an annual rainfall zone of 400–600mm but many enterprises will budget to apply 4–8ML/ha of irrigation water (AgriFutures Australia, 2019). An efficient water management should be practiced to maximise water use efficiency and to avoid waterlogging (Lucerne is intolerant of even short periods of waterlogging) in irrigated Lucerne. Depending on the end use of the crop, the irrigation strategy will be designed to provide adequate moisture at critical development points of the crop hence monitoring of the soil moisture (e.g., using tensiometers) is recommended (AgriFutures Australia, 2019).

Depending on varieties, Lucerne requires a period of dormancy over winter to build up its store of energy for growth in spring and summer. Adequate dormancy is important for the persistence or longevity of the crop, as well as protecting the plant from winter cold (AgriFutures Australia, 2019). Dormant and semi-dormant varieties grow very little during winter, while winter-active varieties grow in winter. It should be noted that all Lucerne varieties grow well during spring and summer, but their growth’ pattern depends on the availability of soil moisture. In general, dormant
and semi-dormant varieties persist longer than highly winter-active ones under stressful conditions (NSW Department of Primary Industries, 2013).

7. Irrigation requirement estimated with FAO-56

Development of efficient irrigation guidelines requires detailed understanding of the irrigation requirement for different crops, considering the effect of the climate regime (Phogat et al., 2019). In the present study, the FAO-56 dual crop coefficient method (Allen et al., 1998, 2005) has been used to estimate the long-term irrigation requirements of different crops (potato, carrot, onion, wine grapes, almond, pistachio, pasture) for current and future climate in different soils [Calcareous (Cal), hard red brown (HRB), sand over clay (SoC) and deep uniform to gradational (DuG)] in the NAP region. Details of the methodology and resulting IR for all soil-crop combinations were discussed in Annex 7. We limited the discussion here to a high-level summary of the findings in terms of cumulative distribution functions (CDF) of annual irrigation requirements (note that this statistical analysis complements the initial study).

Figure 9 to Figure 15 show the CDF of annual irrigation requirement (IR) under current and future climate for different crops (potato, carrot, onion, wine grapes, almond, pistachio, pasture), cultivated on various soils in the NAP region. Results show that crops will require a higher amount of annual irrigation under future climate, depending on the soil textures and crop stress tolerance. Regardless of the crop type, sand over clay soils require the highest irrigation (714.3 – 955.7 mm) while deep uniform to gradational soils need the lowest irrigation (642.8 – 910.0 mm). The average percent increase in IR in different soils ranges from:

- 3.5-5.8% for grapes;
- 6.0-7.2% for almonds;
- 3.0-4.5% for pistachios;
- 7.0-8.4% for pasture;
- 6.2-7.4% for carrot;
- 9.2-10.3% for onion;
- 8.8-11.0% for potato.
This highlights the greater dependency of these crops on irrigation. It should be noted that the salinity of the region’s water resources presents a challenge to some irrigators (not only to irrigators of perennial crops). Some NAP producers of potatoes and almonds currently face significant salinity pressure (particularly those accessing saline groundwater). Increasing water application will also increase the mass of salt imported into the growing systems. To mitigate this problem, savvy irrigation management techniques including aligning leaching events with rain and making use of less saline water sources (recycled water) are recommended.

Figure 9 CDF of annual irrigation requirement for winegrapes under historic and future climate. Cal soil = calcareous soil; HRB soil = hard red brown soil; SoC soil = sand over clay soil; DuG = deep uniform to gradational soil.
Figure 10 CDF of annual irrigation requirement for almonds under historic and future climate. Cal soil = calcareous soil; HRB soil = hard red brown soil; SoC soil = sand over clay soil; DuG = deep uniform to gradational soil.
Figure 11 CDF of annual irrigation requirement for pistachios under historic and future climate. Cal soil = calcareous soil; HRB soil = hard red brown soil; SoC soil = sand over clay soil; DuG = deep uniform to gradational soil.
Figure 12 CDF of annual irrigation requirement for pasture under historic and future climate. Cal soil = calcareous soil; HRB soil = hard red brown soil; SoC soil = sand over clay soil; DuG = deep uniform to gradational soil.
Figure 13 CDF of annual irrigation requirement for carrots under historic and future climate. Cal soil = calcareous soil; HRB soil = hard red brown soil; SoC soil = sand over clay soil; DuG = deep uniform to gradational soil.
Figure 14 CDF of annual irrigation requirement for onions under historic and future climate. Cal soil = calcareous soil; HRB soil = hard red brown soil; SoC soil = sand over clay soil; DuG = deep uniform to gradational soil.
8. Climate change effect on plant water stress

Traditionally readily available water (RAW) is defined as the fraction of total available soil water (TAW) which a crop can extract from the root zone without suffering water stress. This situation occurs when the pressure head is at optimal condition for root water extraction. TAW (mm) is defined as

\[
TAW = 1000 \left( \theta_{fc} - \theta_{wp} \right) Z_r
\]

(1)

where \( \theta_{fc} \) is the water content at field capacity (cm\(^3/cm^3\)), \( \theta_{wp} \) represents the water content at wilting point (cm\(^3/cm^3\)) and \( Z_r \) is the rooting depth (m). The fraction of TAW which a crop can extract from the root zone without suffering water stress is the RAW, given as:

Figure 15 CDF of annual irrigation requirement for potatoes under historic and future climate. Cal soil = calcareous soil; HRB soil = hard red brown soil; SoC soil = sand over clay soil; DuG = deep uniform to gradational soil.
where \( p \) is the average fraction of TAW that can be depleted from the root zone before water stress occurs (0-1). Because both RAW and TAW are static parameters that give no indication of the actual soil water available to crops, an alternative approach was adopted to compare conditions under which a crop experiences crop stress under historic and future climate.

In the present work, the Feddes (1978) model was used to define water stress conditions in soils and its effects on root water uptake, using root water uptake parameters (Feddes parameters) from Table 4. Following Rassam et al. (2018), the Feddes model assumes water uptake to be zero when the soil is close to its saturation state (i.e., wetter than some arbitrary “anaerobiosis point” \( P_0 \)), and in the situations when the pressure head is less (more negative) than the wilting point (\( P_3 \)). For pressure heads between \( P_2 \) and \( P_3 \) (or \( P_0 \) and \( P_{opt} \)), the relationship between water uptake with pressure head is linear. The root water uptake is optimal between pressure heads \( P_{opt} \) and \( P_2 \). Root water uptake can be optimal even at full saturation, when both \( P_0 \) and \( P_{opt} \) are equal to zero.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>( P_0 ) (cm)</td>
<td>-10</td>
</tr>
<tr>
<td>( P_{opt} ) (cm)</td>
<td>-25</td>
</tr>
<tr>
<td>( P_2 ) (cm)</td>
<td>-200</td>
</tr>
<tr>
<td>( P_2 ) (cm)</td>
<td>-800</td>
</tr>
<tr>
<td>( P_3 ) (cm)</td>
<td>-8000</td>
</tr>
<tr>
<td>( r_{2H} ) (cm/day)</td>
<td>0.5</td>
</tr>
<tr>
<td>( r_{2L} ) (cm/day)</td>
<td>0.1</td>
</tr>
</tbody>
</table>

For pasture, the pressure head (\( P_{2L} \)) of 800 cm (or equivalent matric suction of -800 cm) was considered to be the highest suction beyond which root water extraction becomes suboptimal (i.e. stressed). We calculated the number of total days the pasture’s root zone would experience water stress, i.e. number of days with a matric suction was less than -800 cm. This was calculated for both historic and future climate (Table 5). Under future climate, pastures on deep uniform to gradational soils will experience 243 extra water stress days than under historic climate, or 7.6 days per year. Over the 32 years period, hard red brown soils will experience only 105 extra water stress days than under historic climate, or 3.3 days per year.
stress days or 3.3 per year under future climate. Note that the considered water stress is only a practical water stress, as plants can still extract water but it will be under sub-optimal conditions. Full water stress (plant water uptake ceases completely) occurs for pasture at a matric suction of -8000 cm (Table 4).

Table 5 Number of days over the complete simulation period (32 years or 11869 days) that the soil matric suction over the root zone was less than -800 cm. Difference between historic and future climate and average days per year.

<table>
<thead>
<tr>
<th>Soil group</th>
<th>Historic climate</th>
<th>Future climate</th>
<th>Difference</th>
<th>Days/year</th>
</tr>
</thead>
<tbody>
<tr>
<td>Calcareous</td>
<td>8274</td>
<td>8487</td>
<td>213</td>
<td>6.7</td>
</tr>
<tr>
<td>Hard red brown</td>
<td>6397</td>
<td>6502</td>
<td>105</td>
<td>3.3</td>
</tr>
<tr>
<td>Sand over clay</td>
<td>7314</td>
<td>7501</td>
<td>187</td>
<td>5.8</td>
</tr>
<tr>
<td>Deep uniform to gradational</td>
<td>8597</td>
<td>8840</td>
<td>243</td>
<td>7.6</td>
</tr>
</tbody>
</table>

The full data set of simulated root zone pressure heads for pasture for 32 years of simulation (11869 days with each day a data point) is shown as cumulative distribution function (probability of non-exceedance) in Figure 16 to Figure 19. Each graph shows the percentage of the simulation time the soil matric suction was at a certain value. Two threshold values are also shown: the matric pressure P2 (see Table 4) beyond which the water uptake is no longer optimal (here -800 cm), and the matric pressure P3 beyond which the root water uptake ceases completely (here -8000 cm). This allows to estimate the percentage of time the crop has optimal root water uptake, the percentage of time the crop experiences partial water stress (between P2 and P3), and the percentage of time the crop does not take up water (beyond P3: full water stress). The largest difference in soil matric pressure between historic and future climate is observed during the partial water stress period. This is the period during which irrigation becomes most beneficial.
Figure 16 CDF (probability of non-exceedance) of matric suction for pasture subject to a natural condition (only rainfall, no irrigation) under historic and future climate (calcareous soil). Data covers 32 years of simulation.
Figure 17 CDF (probability of non-exceedance) of matric suction for pasture subject to a natural condition (only rainfall, no irrigation) under historic and future climate (hard red brown soil). Data covers 32 years of simulation.
Figure 18 CDF (probability of non-exceedance) of matric suction for pasture subject to a natural condition (only rainfall, no irrigation) under historic and future climate (sand over clay soil). Data covers 32 years of simulation.
9. Conclusions

The effect of climate extremes on crop growth and irrigation requirement in the Northern Adelaide Plains (NAP), South Australia (SA) was investigated using two time series of identical length as representatives of historic (1985-2017) and future (2018-2050) climate. The climate indices for historic and future climate were analyzed to determine the frequency of extreme climate events, and its impact on the growth of horticultural crops (potatoes, carrots, and onions), fruit trees (vines, almonds, and pistachios), and broad acre crops (Lucerne and pasture).

Results show that the NAP region will be subjected to more hot (> 35°C) and dry days (< 1 mm rain), and milder winters. It was predicted that different crops in the region will be impacted...
differently by these climatic shifts. Potatoes will be subjected to a yield decrease and a higher risk to being invaded by pests, while the growth and yield in carrots will be stimulated due to increased frequency of hot and dry days (and dry spells). However, extreme heat events may reduce the quality of carrots and mid-season drought stress depress the yield in carrots. The warmer climate may reduce the duration of crop growth, yield, and seed production for onions, whereas, the low rainfall condition can reduce the risk of infection by pests (i.e., leaf blight). The projected drought and extremely hot weather in the NAP region can negatively impact vines, which may results in poor budburst, leaf loss, bunch damage, and consequently low yield and production or even crop loss. Warmer summers may lead to adopting warmer climate grape varieties to avoid loss of quality of wines. For almonds and pistachios, it is expected that their yield and production will be impacted by drought in the NAP region if not properly managed. While the current NAP climate hardly accommodates chill requirements for these fruits, the projected climate show there would be some years that the chilling requirement cannot be met.

As the NAP region will be subjected to an increased number of hot and dry days, higher irrigation requirements are needed. Irrigation requirements for abovementioned crops, simulated via the FAO-56 dual crop coefficient method, revealed that generally crops will require a higher amount of annual irrigation in the future, depending on the soil textures and crop stress tolerance. Simulation results illustrated that sand over clay soils require the highest irrigation (from 714 to 956 mm) while deep uniform to gradational soils need the lowest irrigation (from 643 to 910 mm) among other soils, regardless of the crop type. It is anticipated that annual horticultural crops face more irrigation related risks under the future climate compared to deep rooted perennial horticultural crops.

The results of water balance simulation showed that under future climate, pastures on deep uniform to gradational soils will experience 243 extra water stress days (over a 32 year period) compared to historic climate, or 7.6 days per year. Over the 32 years period, hard red brown soils will experience only 105 extra water stress days or 3.3 per year under future climate. This highlights the fact that under future climate, pasture’s production will be reduced in the absence of irrigation.

As the NAP region will be subjected to increased number of hot days, a decline in pasture production is anticipated for this region. This is in agreement with previous studies (Cullen et al.)
which predicted that climate change will lead to decline in pasture production in South Australia.

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