Impacts of Amelioration on Sodic Soil Hydraulic Conductivity and Potential Consequences for Deep Drainage in the Lower Burdekin, North Queensland

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School of Engineering
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My supervisors, Professor David Lockington, Dr Keith Bristow and Dr Thomas Baumgartl each contributed to editing the Thesis prior to submission.

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“None.”
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Abstract
Groundwater tables are rising beneath irrigated fields in the Lower Burdekin in North Queensland, Australia. The soils where this occurs are predominantly sodic clay soils with low hydraulic conductivities. Many of these soils have been treated by applying gypsum or by increasing the salinity of irrigation water by mixing saline groundwater with fresh river water. While the purpose of these treatments is to increase infiltration into the surface soils and improve productivity of the root zone, it is thought that the treatments may have altered the soil hydraulic properties well below the root zone leading to increased groundwater recharge and rising water tables.

The objective of this thesis is to assess the potential for sodic soil amelioration treatments to increase soil hydraulic conductivity and hence increase groundwater recharge rates. This research focuses on the impacts of gypsum treatments and increased irrigation water salinity on soil chemistry, soil hydraulic conductivity and deep drainage rates.

To determine the current properties of local sodic soils, previous soil surveys were reviewed and field work was conducted to collect soil samples. From the analysis of these soils, it was determined that sodicity levels remain high at some locations even under irrigation and that the high sodicity levels extend below the typically sampled “root zone”

Using the samples of the collected sodic soils, experiments were conducted to measure changes in hydraulic conductivity in response to selected treatments. The results of these column experiments showed that gypsum applications can lead to increases in hydraulic conductivity of an order of magnitude when compared with the maximum hydraulic conductivity of a surrogate for good quality irrigation water. Hydraulic conductivity increases occurred when sodic soils were leached continuously with a saturated gypsum solution for at least three weeks. These increases in hydraulic conductivity were positively correlated with decreases in exchangeable sodium and magnesium concentrations. The salt concentration of the applied solution was found to have a significant influence on the maximum hydraulic conductivity; however continued leaching with mixed cation saline waters caused a rapid increase in hydraulic conductivity followed by a gradual decrease.
Numerical modelling was used to study the expected depth and timing of the impacts of amelioration, compare the impacts of different treatments on deep drainage and to determine the influence of large amounts of rain during the relatively short wet season on soil properties and deep drainage. When the amelioration of ten metre profiles of sodic clays was simulated using HYDRUS-1D, decreases in soil sodicity only occurred close to the soil surface but deep drainage rates still increased. When gypsum applications occurred during the irrigation season, deep drainage rates during the following wet season were at least three times greater than when no gypsum was applied.

Gypsum applications and increased irrigation salinity have the potential to increase soil hydraulic conductivity and deep drainage rates. The overall impact of sodic soil amelioration on rising groundwater tables in the Lower Burdekin depends on the initial soil sodicity and soil hydraulic properties of the active crop root zone, the frequency and duration of treatments, the rates of irrigation and rainfall, and the hydraulic conductivity of the materials in the deeper unsaturated zone below the crop root zone.

Keywords
gypsum, sodic soils, amelioration, hydraulic conductivity, deep drainage, groundwater recharge

Australian and New Zealand Standard Research Classifications (ANZSRC)
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<tbody>
<tr>
<td>Na</td>
<td>Sodium</td>
</tr>
<tr>
<td>Ca</td>
<td>Calcium</td>
</tr>
<tr>
<td>Mg</td>
<td>Magnesium</td>
</tr>
<tr>
<td>K</td>
<td>Potassium</td>
</tr>
<tr>
<td>Cl</td>
<td>Chloride</td>
</tr>
<tr>
<td>HCO$_3$</td>
<td>Bicarbonate</td>
</tr>
<tr>
<td>SO$_4$</td>
<td>Sulphate</td>
</tr>
<tr>
<td>CaSO$_4$</td>
<td>Calcium sulphate (gypsum)</td>
</tr>
<tr>
<td>CaCO$_3$</td>
<td>Calcium Carbonate</td>
</tr>
<tr>
<td>H$_4$SiO$_4$</td>
<td>Silica</td>
</tr>
<tr>
<td>NH$_4$Cl</td>
<td>Ammonium chloride</td>
</tr>
<tr>
<td>KNO$_3$</td>
<td>Potassium nitrate</td>
</tr>
<tr>
<td>CO$_2$</td>
<td>Carbon dioxide</td>
</tr>
<tr>
<td>CO$_3$</td>
<td>Carbonate</td>
</tr>
<tr>
<td>CO$_3^T$</td>
<td>Total carbonates</td>
</tr>
<tr>
<td>HCO$_3^T$</td>
<td>Total bicarbonates</td>
</tr>
<tr>
<td>[OH$^-$]</td>
<td>Hydroxide ion</td>
</tr>
<tr>
<td>[H$^+$]</td>
<td>Hydrogen ion</td>
</tr>
<tr>
<td>pH</td>
<td>Negative logarithm of the hydrogen ion activity in a solution</td>
</tr>
<tr>
<td>Ug</td>
<td>Cracking clay soil</td>
</tr>
<tr>
<td>K</td>
<td>Hydraulic conductivity [L/T]</td>
</tr>
<tr>
<td>Q</td>
<td>Flow rate [L$^3$/T]</td>
</tr>
<tr>
<td>L</td>
<td>Length of Column [L]</td>
</tr>
<tr>
<td>A</td>
<td>Cross-sectional Area of Column [L$^2$]</td>
</tr>
<tr>
<td>h</td>
<td>Water pressure head [L]</td>
</tr>
<tr>
<td>$\theta$</td>
<td>Volumetric water content [L$^3$/L$^{-3}$]</td>
</tr>
<tr>
<td>z</td>
<td>Spatial coordinate [L]</td>
</tr>
<tr>
<td>S</td>
<td>Sink term [L$^3$L$^{-3}$T$^{-1}$]</td>
</tr>
<tr>
<td>$\phi$</td>
<td>The angle between the flow direction and the vertical axis</td>
</tr>
<tr>
<td>$\theta_s$</td>
<td>Saturated water content [L$^3$L$^{-3}$]</td>
</tr>
</tbody>
</table>
\[ \theta_r \] Residual water content \([L^3L^{-3}]\)
\[ \theta_m \] Extrapolated parameter - slightly larger than \(\theta_s\) \([L^3L^{-3}]\)
\[ \theta_a \] Extrapolated parameter (less than or equal to \(\theta_r\)) \([L^3L^{-3}]\)
\[ \theta_k \] Extrapolated parameter (less than or equal to \(\theta_s\)) [-]
\[ \alpha, m, n \] Empirical parameters in the van Genuchten (1980) equation \([1/L], [-], [-]\)
\[ l \] Pore connectivity parameter
\[ S_e \] Effective water content [-]
\[ K_s \] Saturated hydraulic conductivity \([L/T]\)
\[ c_{Ti} \] Total dissolved concentration of the aqueous component \(i\) \([ML^{-3}]\)
\[ \bar{c}_{Ti} \] Total exchangeable concentration of the aqueous component \(i\) \([MM^{-1}]\)
\[ \hat{c}_{Ti} \] Total non-adsorbed solid phase concentration of aqueous component \(i\) \([MM^{-1}]\)
\[ \rho \] Soil bulk density \([ML^{-3}]\)
\[ D \] Dispersion coefficient \([L^2T^{-1}]\)
\[ q \] Volumetric flux \([LT^{-1}]\)
\[ K_r \] Relative hydraulic conductivity \([L/T]\)
\[ r \] Hydraulic conductivity scaling parameter (McNeal 1968)
\[ r_1 \] Scaling parameter representing the effect of the soil solution pH on hydraulic conductivity
\[ r_2 \] Scaling parameter representing the effect of ESP (SAR) and soil solution salinity on the hydraulic conductivity
\[ K(h) \] Hydraulic conductivity corresponding to a local pressure head \([L/T]\)
\[ y \] Relative soil hydraulic conductivity
\[ x \] A swelling factor
\[ c, n \] Constants for a given soil within a specified range of ESP values
\[ f_{mont} \] Weight fraction of montmorillonite in the soil
\[ ESP^* \] Adjusted ESP
\[ d^* \] Adjusted interlayer spacing
\[ C_0 \] Total salt concentration of the ambient solution
\[ \alpha \] Dispersivity \([L]\)
\[ |q| \] Darcy water flux \([LT^{-1}]\)
\[ Dm \] Molecular diffusion coefficient \([L^2T^{-1}]\)
\[ \tau \] Tortuosity [-]
\[ K_{ij} \] Cation exchange coefficient for exchange between species i and j
LIST OF ACRONYMS AND ABBREVIATIONS

AAS Atomic Absorption Spectroscopy
AHD Australian Height Datum
BD Bulk Density
BHWSS Burdekin Haughton Water Supply Scheme
CEC Cation Exchange Capacity
EC Electrical Conductivity
ESP Exchangeable Sodium Percentage
meq Milli equivalents
NATA National Association of Testing Authorities
SALI Soil and Land Information
SAR Sodium Adsorption Ratio
USDA United States Department of Agriculture
XRD X-ray Diffraction
### GLOSSARY

<table>
<thead>
<tr>
<th>Term</th>
<th>Definition</th>
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<tbody>
<tr>
<td>Alkali soils</td>
<td>Soils with a pH of 8.5 or higher or with a exchangeable sodium percentage greater than 15.</td>
</tr>
<tr>
<td>Amelioration</td>
<td>The act of making better, improvement of conditions.</td>
</tr>
<tr>
<td>Clay mineral</td>
<td>Any of a group of important hydrous aluminium silicates with a layered structure and very small (less than 0.005 nm) particle size.</td>
</tr>
<tr>
<td>Diffuse double layer</td>
<td>A system that consists of a solid surface (e.g. clay surface) having a net electrical charge together with an ionic swarm in solution containing ions of opposite charge, neutralizing the surface charge.</td>
</tr>
<tr>
<td>Exchangeable cation</td>
<td>A positively charged ion held on or near the surface of a solid particle by a negative surface and which may be replaced by other positively charged ions in the soil solution.</td>
</tr>
<tr>
<td>Gypsum</td>
<td>Calcium sulphate (CaSO$_4$·2H$_2$O), used to ameliorate soils with a high exchangeable sodium percentage.</td>
</tr>
<tr>
<td>Hydraulic conductivity</td>
<td>The proportionality factor in Darcy’s law as applied to viscous flow of water in soil, i.e. the flux of water per unit gradient of hydraulic potential.</td>
</tr>
<tr>
<td>Kaolinite</td>
<td>A 1:1 layer silicate clay mineral.</td>
</tr>
<tr>
<td>Lime</td>
<td>Agricultural lime is a soil amendment containing calcium carbonate, used to neutralize soil acidity and furnish calcium and magnesium for plant growth.</td>
</tr>
<tr>
<td>Leachate</td>
<td>Liquids that have percolated through a soil and that contain substances in solution or suspension.</td>
</tr>
<tr>
<td>Reclamation</td>
<td>The act of reclaiming, restoration.</td>
</tr>
<tr>
<td>Term</td>
<td>Definition</td>
</tr>
<tr>
<td>--------------</td>
<td>--------------------------------------------------------------------------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Saline soils</td>
<td>Saline soils are those which have an electrical conductivity of the saturation soil extract of more than 4 dS/m (USDA 1954).</td>
</tr>
<tr>
<td>Sesquioxides</td>
<td>A general term for oxides and hydroxides of iron and aluminium.</td>
</tr>
<tr>
<td>Shandy</td>
<td>Mixture of groundwater and surface water used for irrigation.</td>
</tr>
<tr>
<td>Smectite</td>
<td>A group of 2:1 layer silicate clay minerals that includes montmorillonite.</td>
</tr>
<tr>
<td>Sodic soils</td>
<td>Sodic soils have an exchangeable sodium percentage of greater than 15 based on the American definition (USDA 1954) or greater than 6 based on the Australian definition (Northcote and Skene 1972).</td>
</tr>
<tr>
<td>Tactoid</td>
<td>The colloidal sized aggregates of phyllosilicate clay minerals that can form under certain conditions of exchangeable cations and ionic strength.</td>
</tr>
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</table>
CHAPTER 1 – INTRODUCTION

Sustainability of irrigated agriculture is threatened by a combination of several interrelated factors, including lack of fresh water, lack of drainage, shallow groundwater tables and salinization of soil and groundwater resources (Schoups, Hopmans et al. 2005). The extent of salt affected soils globally has been estimated to cover more than 950 million ha (Lal and Stewart 1990). In Europe and Australia, a large proportion of these salt affected soils are alkali soils (Szabolcs 1989) and many of these soils are sodic. In Australia, over 80% of irrigated soils are sodic and irrigation management is closely linked with management of soil sodicity (Rengasamy and Olsson 1993).

1.1 Sodic soils

The terms “saline” and “sodic” are often confused. While saline soils are those which have an electrical conductivity of the saturation soil extract of more than 4 dS/m (USDA 1954), sodic soils have an exchangeable sodium percentage of greater than 15 based on the American definition (USDA 1954) or greater than 6 based on the Australian definition (Northcote and Skene 1972). Salinity in Australia is dominated by sodium chloride salt (Rengasamy and Olsson 1993) and as such, it is common for saline soils in Australia to be sodic.

Sodic soils typically exhibit poor soil structural stability, low plant available water contents and low infiltration rates (Nelson, Ham et al. 2001; Shainberg and Letey 1984; Shaw, Brebber et al. 1994). They have a tendency to disperse when wet and become hard setting when dry (Nelson, Lawer et al. 2002). Clay swelling and clay dispersion are the key processes responsible for all of these properties.

The processes of swelling and dispersion are the result of the interaction between the constituents of the clay fraction and the solution composition within pores (Sumner 1993).
In clays, diffuse electrical double layers are made up of clay lattice charges and a sea of compensating counter ions (Shainberg and Letey 1984). The interactions that occur at this scale are described by diffuse double layer theory.

As the thickness of the diffuse double layer increases, the repulsive force between clay particles also increases leading to increased swelling pressures and an increased tendency for clay dispersion (Sumner 1993). Clay swelling in layered silicate clays occurs due to the expansion of the clay lattice when water is incorporated into the lattice structure, between the clay layers (Moyne and Murad 2003). When the salt concentration is lowered, swelling pressures are further increased as more water is incorporated into the clay lattice by osmosis (Moyne and Murad 2003).

The presence of sodium ions increases the thickness of the diffuse double layers, thereby increasing the distances between the clay particles, increasing clay swelling and promoting dispersion (Sumner 1993). When sodic soils are also saline, the presence of high concentrations of ions in between clay platelets reduces the thickness of the diffuse double layer and helps to stabilize the soil. However, when the soil is leached with low salinity water, the stability of the soil is reduced and clay dispersion occurs (Shainberg, Rhoades et al. 1981a).

For montmorillonites, the differences in clay platelet structure between sodium saturated clays and calcium saturated clays have been well established. In sodium montmorillonites, single platelets tend to persist in solution (Shainberg and Letey 1984). However, in calcium montmorillonites, the platelets aggregate into “tactoids” (Blackmore and Miller 1961) or “quasi crystals” (Aylmore and Quirk 1959). Each quasi crystal consists of four to nine platelets in a parallel array (Shainberg and Letey 1984). This stacking of clay platelets increases the volume of pore space in calcium saturated clays.

Our understanding of the response of sodic soils to changes in soil chemistry comes mostly from laboratory experiments. The effects of electrolyte concentration and soil sodicity on clay swelling and soil hydraulic properties have been studied by a number of authors (Aylmore and Quirk 1959; Gardner, Mayhugh et al. 1959; McNeal and Coleman 1966; Quirk and Schofield 1955). The work of McNeal resulted in functions relating soil salinity
and sodicity to hydraulic conductivity (McNeal 1968). Although these functions have been used to predict changes in hydraulic conductivity for a limited number of soils, differences in responses between soils mean that laboratory measurements are still needed to enable these functions to be applied to specific soils (McNeal 1968).

Suarez, Rhoades et al. (1984) conducted experiments that showed that the pH of the soil solution also affects hydraulic conductivity. It has been suggested that the reason that hydraulic conductivity increases with decreasing pH is because with decreases in pH there are increases in the negative edge charge on clay surfaces and the positive charge of iron and magnesium oxides, leading to increased bonding between clay particles and decreased clay dispersion (Suarez, Rhoades et al. 1984). pH effects on swelling are particularly important for kaolinites and sesquioxides (Sumner 1993)

Some of the soil properties that have been found to influence the response of hydraulic properties to changes in soil chemistry include organic matter content and the presence of oxides (Churchman, Skjemstad et al. 1993), bulk clay content and clay mineralogy (Frenkel, Goertzen et al. 1978) and mineral weathering (Shainberg, Rhoades et al. 1981b). While it is not possible to predict reductions in hydraulic conductivity of a soil based on knowledge of its dominant clay mineralogy alone (Churchman, Skjemstad et al. 1993), clay mineralogy does provide a qualitative insight into the reductions in hydraulic conductivity that might be expected for soils containing certain clay minerals (McNeal and Coleman 1966). For cracking clays soils, soils with high clay contents and smectite mineralogy were found to be less susceptible to decreases in hydraulic conductivity at a given sodicity than soils of moderate clay content and mixed mineralogy (Shaw, Brebber et al. 1994). As there are a number of properties that influence how a given soil will respond to changes in soil chemistry, laboratory experiments need to be conducted for soils of interest.
1.2 Amelioration of Sodic Soils

Most of the irrigated soils in Australia need reclamation of sodicity, at least for the soil layers in the root zone (Rengasamy and Olsson 1993). The aim of sodic soil amelioration is to improve soil properties by reducing interlayer swelling. This can be achieved by reducing the proportion of sodium on the exchange complex or increasing the salinity of the soil solution.

Gypsum (Calcium Sulphate, CaSO\(_4\)) is the most commonly used amendment for sodic soil reclamation, primarily due to its low cost (Shainberg, Keren et al. 1982). It works by replacing exchangeable sodium ions with the applied calcium ions and by increasing the salt concentration of the soil solution. These two mechanisms are referred to respectively as cation exchange and electrolyte effects (Loveday 1976). The result is reduced dispersion, increased infiltration rates and increased hydraulic conductivities. Field studies have measured impacts of gypsum treatments on sodic soils including increased rates of water entry (Davidson and Quirk 1961), decreases in exchangeable sodium concentration (Greene and Ford 1985) and increased crop yields (McKenzie and So 1989).

Some of the alternative treatment methods that are used for sodic soils include agricultural lime (Calcium Carbonate, CaCO\(_3\)) or saline irrigation waters. Lime is sometimes used as a treatment for acidic sodic soils (Nelson, Ham et al. 2001). It is not suitable for alkaline sodic soils and is much less soluble than gypsum. Saline irrigation waters that are sometimes used include wastewaters or saline groundwaters. The composition of these waters needs to be considered as there is a risk of increasing soil sodicity if the waters have high concentrations of sodium relative to calcium, magnesium and potassium.

1.3 Risk of Rising Groundwater Tables

When soils are successfully ameliorated, the leaching flux is expected to increase relative to surface runoff potentially increasing deep drainage rates (Rengasamy and Olsson 1993).
The risk of groundwater recharge rates increasing as a result of sodic soil amelioration is contributed to by a number of factors. These factors include initial soil sodicity, initial soil hydraulic properties, efficacy of the treatments, duration and frequency of the treatments, rainfall and irrigation rates and evapotranspiration rates. The greater the depth of the profile that is affected by the amelioration treatments, the higher the likelihood that the increases in infiltration rates will result in increases in groundwater recharge rates.

1.4 Sodic Soil Management in the Lower Burdekin

The Burdekin River catchment in North East Queensland covers an area of approximately 136,000 km$^2$ (Beare, Bell et al. 2003). The floodplains of the Burdekin catchment, adjacent to the Great Barrier Reef Lagoon, are referred to as the “Lower Burdekin”. The Lower Burdekin is one of Queensland’s premier irrigation areas and has a reputation for producing some of the highest sugarcane yields in Australia (Bristow, Charlesworth et al. 2003). The success of sugarcane production in the region has been attributed to the suitability of soils and climate and the availability of water supplies (Bristow, Charlesworth et al. 2003).

The development of large areas of sodic soils for irrigation has occurred in the last fifteen years. Early field experiments on local sodic soils showed improvements in sugarcane yields (McMahon, Ham et al. 1996) and increases in infiltration rates (Dowling and Elliot 1988) when gypsum was applied. Gypsum treatments have since been used widely to treat these soils (McMahon, Ham et al. 1996). In some areas, saline groundwater is mixed with the river water used for irrigation, as an alternative treatment. The potential for wide-scale amelioration of these soils to contribute to rising groundwater levels as a result of increases in infiltration rates is a concern.

In the presence of high clay contents in sodic soils and deep clay profiles, it is possible that the impacts of sodic soil amelioration may be dampened. The stratigraphy beneath irrigated fields in the Lower Burdekin includes sand and clay lenses. In some places, clayey materials extend from the soil surface to depths in excess of ten metres. The sodic
soils at the surface typically have high sodicity levels of above 15 and high clay contents of up to 70 percent. To determine the potential impacts of sodic soil amelioration in the Lower Burdekin, measurements and modelling are needed.

While there have been measurements made previously to determine the impacts of applied solution chemistry on soil hydraulic properties, for example McNeal and Coleman (1966) and Suarez, Rhoades et al. (1984), the effects are soil specific so laboratory measurements are needed to determine the relationships applicable for the sodic soils from the Lower Burdekin. To predict the impacts of sodic soil amelioration on deep drainage rates beneath irrigated fields in the Lower Burdekin, modelling is needed.

1.5 Research Questions

The research questions addressed in this thesis include:

- What are the current sodicity levels of sodic soils in the Lower Burdekin?
- Are sodicity levels high below the root zone?
- How much influence do gypsum applications and increased salinity waters have on the hydraulic conductivity of sodic soils from the Lower Burdekin?
- What is the likelihood that gypsum applications have increased deep drainage rates?
- What is the expected depth of reductions in soil sodicity resulting from different gypsum application frequencies?
- How would seasonal flushing by rainfall influence reductions in soil sodicity?

1.6 Approach

To address the research questions listed above, an approach was devised that includes soil sampling, laboratory experiments and numerical modelling. Pre-development soils mapping was used to help select sites for the soil sampling. The soils were then used in
laboratory experiments to assess the changes in hydraulic conductivity that result from continuous applications of gypsum and increased salinity waters, by continuing to leach the soils until flow rates stabilized. Changes in soil chemistry during leaching were also assessed during these experiments.

Numerical modelling was used to simulate the impacts of sodic soil amelioration on soil hydraulic properties and deep drainage rates. HYDRUS (Simunek, van Genuchten et al. 2005) was chosen as the most suitable tool for modelling the amelioration of sodic soils as it enables the simulation of reductions in hydraulic conductivity that can occur in sodic soils, in addition to saturated and unsaturated water flow and reactive solute transport.

The UNSATCHEM model, which has now been incorporated into HYDRUS as an optional module, has been used previously to model sodic soil amelioration (Simunek and Suarez 1997). Replication of these published model results was used as a starting point for the modelling in this thesis. This was followed by simulations using the properties of sodic soils from the Lower Burdekin. The models were then extended to simulate deeper soil profiles for longer time periods. Finally, models were set up to determine the effects of seasonal rainfall, varied gypsum application frequencies and application of gypsum within irrigation furrows.

1.7 Organisation of Thesis

Following on from this introduction in Chapter 1, the setting for this study is described in Chapter 2. This Chapter includes a description of irrigation development in the Lower Burdekin, the local climate and details of previous soil surveys. Soil sampling and analyses were used to characterise local sodic soils. The results of this work are outlined in Chapter 2.

In Chapter 3, two sets of column experiments using sodic soils from the Lower Burdekin are described. The purpose of these experiments was to determine the response of local sodic soils to gypsum applications and varied irrigation water salinity. In the first set of
experiments, saturated gypsum solutions were continuously applied to soil columns for several weeks and changes in leachate chemistry and hydraulic conductivity were measured. For this set of experiments, two different bulk densities were used. In the second set of experiments, mixed cation solutions of two different salinities were used as the applied solution. Leaching was continued for several months to allow flow rates to stabilize. The results of these two sets of experiments are discussed and comparisons with previous studies are made.

In Chapter 4, the application of HYDRUS-1D for simulating the amelioration of sodic soils is tested and the short column results of Chapter 3 are modelled. The purpose of these models is to investigate the key processes relating to sodic soil amelioration using the HYDRUS-1D model and to verify whether the general trends observed in the laboratory could be simulated.

Chapter 5 shows how HYDRUS modelling was then extended to study the potential impacts of sodic soil amelioration on deep drainage. Extended HYDRUS-1D models are used to determine the depth of impacts of sodic soil amelioration. These models are also used to assess the impacts of soil properties and different amelioration treatments on deep drainage. Subsequent models investigate the impacts of seasonal rainfall and irrigation, varied gypsum application frequencies and gypsum application within irrigation furrows.

Chapter 6 presents a summary of the conclusions from this thesis. It also discusses some of the implications for the key findings and provides recommendations for further research.
CHAPTER 2 – SODIC SOIL MANAGEMENT AND GROUNDWATER HYDROLOGY IN THE BURDEKIN HAUGHTON WATER SUPPLY SCHEME

2.1 Introduction

Irrigation development commenced in the Burdekin Haughton Water Supply Scheme (BHWSS) in 1949. Groundwater monitoring data shows that groundwater levels have generally been rising below the irrigated soils in the BHWSS. The soils in this area are primarily sodic clay soils and it is possible that amelioration of these sodic soils has contributed to rising groundwater tables. The most common form of sodic soil amelioration in the BHWSS is by gypsum applications and large areas of sodic soils have been treated using this method.

This Chapter introduces the hypothesis that amelioration of sodic soils could be contributing to rising groundwater levels in the BHWSS. To investigate the possibility of the widespread amelioration of sodic soils in the BHWSS contributing to rising groundwater tables, more information was needed about the physical and chemical soil properties that influence soil hydraulic properties. Suitable sites were needed where these soil properties could be determined and samples could be collected for laboratory investigations.

Prior to irrigation development, soil surveys were conducted which resulted in detailed soils maps for parts of the BHWSS. Some of these surveys included measurement of soil chemical properties and resulted in soil sodicity maps. The maps of soil types and soil sodicity were used to identify soil suitability for irrigation and areas where amelioration was needed. The existence of detailed soils maps and soil analyses in the BHWSS
provides a good starting point for investigating the impacts of sodic soil amelioration on soil properties and deep drainage rates below these soils.

Characterisation of the sodic soils in the BHWSS was achieved through a review of previous soil survey data and further soil sampling and analysis. The process involved in selecting field sites for investigation is outlined in this Chapter and results of recent soil analyses are discussed.

2.2 Setting: The Lower Burdekin

The Burdekin River catchment is the second largest river basin on Queensland’s east coast (Figure 2-1). The interior section of the catchment is over 700 kilometres long in the north-south direction and includes the Burdekin, Belyando, Suttor, Bowen, Cape and Campaspe rivers. Major industries in the interior of the Burdekin catchment include mining and grazing.
The Lower Burdekin is the coastal section of the catchments to the south of Townsville which includes the Burdekin and Haughton Rivers and the major towns Ayr, Home Hill, Clare and Giru (Figure 2-2). The region is dominated by irrigated land referred to as the
Burdekin-Haughton Water Supply Scheme (BHWSS) on the western side of the Lower Burdekin and the North and South Burdekin Water Board areas on the coastal side.

The Lower Burdekin is a Delta which has been formed mainly by the changing course of the Burdekin River, resulting in a series of discrete lobes (Fielding, Trueman et al. 2006). Figure 2-3 shows the sequence of sediment deposition as a result of the changing position of the Burdekin River, with early deposition in the west and more recent deposition in the east. Marine deposition has also occurred in the Delta, leading to a complex relationship between terrestrial and marine sediments which is related to the relative position of the coastal boundary over geological time (McMahon, Arunakumaren et al. 2000). The sediments deposited in the Delta, which include sands, gravels, silt, clay and organic muds overlie a granite basement and consist of thicknesses of up to 100 metres near the coast (McMahon, Arunakumaren et al. 2002). In the BHWSS area of the Lower Burdekin, in the west, there is a more extensive coverage of semi-confining surficial clays compared with the coastal section of the Delta (KBR 2002; McMahon, Arunakumaren et al. 2000).
The basement rock beneath the BHWSS consists largely of granitic rocks (KBR 2002). The basement contours indicate a relatively broad and shallow valley through which a network of major palaeochannels has incised into the bedrock (KBR 2002). The bedrock falls away quite steeply from the solid outcrops but a rock bar exists directly across the present course of the Burdekin (Hopley 1970).

The climate of the Lower Burdekin is characterised by a short summer wet season with heavy rainfall, a long winter dry season and moderate to high temperatures (Hubble and Thompson 1953). Average monthly evaporation rates range from 100 to 250 mm and the majority of the annual rainfall falls between December and March (Figure 2-4). The average monthly rainfall and evaporation at the Clare rainfall station was calculated using data downloaded from the Bureau of Meteorology’s SILO website (http://www.bom.gov.au/silo/). The data used was a ‘patched point’ dataset. This data format includes measured data from the metrological station as well as interpolated data to...
fill any gaps in the record. The interpolation methods used are described by Jeffrey, Carter et al. (2001).

![Figure 2-4. Monthly average rainfall and evaporation at Clare based on data from 1900 to 2006.](image)

The high incidence of summer rainfall in the Lower Burdekin coupled with high evaporation rates results in irrigation being necessary for year-round crop production. The success of sugarcane production in the region has been attributed to the suitability of soils and climate and the availability of water for irrigation (Bristow, Charlesworth et al. 2003).

The dominant irrigation method is furrow irrigation (Qureshi, Harrison et al. 2002). This form of irrigation is favoured where slopes are less than 3%. The efficiency of furrow irrigation typically ranges from 30% to 90% (Qureshi, Harrison et al. 2002). Although sugarcane is the dominant crop grown, other crops like mangoes and vegetables are also grown (Beare, Bell et al. 2003). The area of irrigated sugarcane and other crops is around 80,000 ha (Bristow, Charlesworth et al. 2001).
2.2.1 Irrigation Development History

Production of sugarcane began in the Burdekin Delta in 1883 (Bristow, Charlesworth et al. 2000). By the end of the nineteenth century, extraction of groundwater from the sandy aquifers of the Delta for irrigation water supplies had become widespread (O'Shea 1985). The farms developed in the Delta were quite small as the infiltration rates were high so furrow lengths needed to be kept relatively short to prevent excessive volumes of irrigation water from needing to be applied.

Enhanced extraction combined with decreased recharge (from rainfall) and the risk of sea water intrusion resulted in the establishment of an artificial recharge scheme in 1965. This scheme was managed by the newly established water boards, the North Burdekin Water Board and the South Burdekin Water Board which were established in 1965 and 1966 respectively.

Irrigated agriculture in the Burdekin Haughton Water Supply Scheme (BHWSS) area adjacent to the Burdekin Delta (Figure 2-2) began with the irrigation of tobacco crops in the Clare section in 1949 (Petheram, Charlesworth et al. 2006). Mona Park was the next section to be developed for irrigation in 1965 but unlike Clare, it was irrigated with groundwater instead of river water.

Further development was then planned for additional sites on the left and right banks of the Burdekin River. Release of farms on the left bank was progressed more quickly than farms on the right bank due to difficulties in managing salinity outbreaks and rising groundwater levels on the right bank (QWRC 1985). It was initially believed that there would be less risk of these issues being a problem on the left bank.

The use of surface water for irrigating crops increased with the establishment of the Burdekin Falls Dam in 1987 and the construction of surface water supply channels in the BHWSS. A conjunctive use policy was recommended for the BHWSS area to help limit rising groundwater levels which have the potential to lead to salinity problems.
Irrigation development in the BHWSS was partially limited by the presence of large areas of sodic soils. Sodic soils, by definition, have a high Exchangeable Sodium Percentage (ESP), which is the percentage of the cation exchange complex occupied by sodium ions. In Australia, soils are classified as sodic if the ESP is greater than six (Northcote and Skene 1972). Characteristic properties of sodic soils include a tendency to disperse when wet and become hard setting when dry (Nelson, Lawer et al. 2002). Once sodic soils were identified as a potential limitation to irrigation development in the BHWSS, intensive soil surveys and soil analyses were performed to determine the distribution and sodicity levels of these soils.

2.2.2 Sodic Soils in the BHWSS

Soils information for the Lower Burdekin is available in the form of maps and technical reports, produced mainly by the Department of Primary Industries (DPI) though some of the early soil surveys were undertaken by CSIRO. Hubble and Thompson (1953) mapped the Lower Burdekin Valley at a scale of about 1:190 000 scale using soil associations and miscellaneous land types as mapping units. A number of soil series were identified and some of the common soil series names are still being used (Donnollan 1991). According to Hubble and Thompson (1953), soils of the Oakey, Dowie and Barratta series were identified as being of doubtful suitability for irrigation. These soils cover a considerable proportion of the BHWSS that have been developed more recently including the Mulgrave section.

More recent soil surveys were then conducted by the Department of Primary Industries. These surveys were based on the subdivision of the primary profile form of Northcote (1979), where soils are grouped by landscape unit and soil types. The first of these surveys included that of Reid and Baker (1984) which was performed at 1:100 000 scale, in order to provide more detailed information on the irrigation potential of the area.

Prior to the subdivision and release of farms for irrigation in the BHWSS, high intensity surveys (1:25 000 scale) were conducted by the DPI (Day 1993; Day 1994; Donnollan 1994a; Donnollan 1994b; Donnollan, McClurg et al. 1986; Loi, Christianos et al. 1994;
Loi and McClurg 1994; McClurg 1995; McClurg, Tucker et al. 1988). These surveys were conducted in the different sections of the BHWSS that were scheduled to be developed (Figure 2-5). Figure 2-6 shows the detailed soils mapping for the Mulgrave section, the majority of the Mulgrave section map is shown and farm boundaries have been included. This series of soils maps provide the best available soils mapping in the BHWSS.

Reports presenting the salinity and sodicity levels for the soil profile classes in the BHWSS were compiled in the early stages of development (Ahern and Weinand 1989; Ahern, Weinand et al. 1989). Soils with a sodic B horizon (ESP > 6) were found to be very extensive within the BHWSS (Day, Loi et al. 1992). In these soils, high levels of subsoil salinity and sodicity were predicted to create problems under irrigation due to exposure of sodic subsoils and water stress due to limited infiltration (Dowling and Elliot 1988).

The most common soil types in the BHWSS are cracking clays and duplex soils, these two soil groups occupy over three quarters or 95 000 ha of the area (Donnollan 1991). High sodicity levels have been measured for cracking clays and duplex soils in the Mulgrave section of the BHWSS (McClurg, Tucker et al. 1988) and these two soil types occupy the vast majority of the Mulgrave section (Figure 2-6).

Cracking clays are those soils which contain more than 35% clay throughout the profile and crack when dry and swell when wet. The phenomena of swelling and shrinking are related to the nature of the clay minerals. Under irrigation, cracking in these soils is expected to be substantially reduced (Weaver, Hulugalle et al. 2005). “Cracking clays” include the soils previously grouped in the Barratta soil series. The texture of the A horizon typically ranges from light to medium clay and the texture of the B horizon ranges from medium to heavy clay (Donnollan 1991).

Duplex soils are those soils which have a distinct change in texture from the A horizon to the B horizon. The duplex soils are divided into subdivisions based on the colour of the B horizon. Common colours in the BHWSS are yellow-grey, brown, red and dark. “Sodic duplex soils” include the Dowie and Oakey soil series. The textures of the A horizon typically vary from sandy loam to clay loam and the textures of the B horizon vary from light to medium clay (Donnollan 1991).
The Mulgrave section on the left bank of the Burdekin River (Figure 2-6) has the greatest area of sodic soils and this could be due to the close proximity to undulating rises on acid intrusive rocks (Day, Loi et al. 1992). The sodicity of soils have been mapped in the Mulgrave section at a number of different depths (Ahern, Rosenthal et al. 1988). At 0.3 m, 70% of soils in the Mulgrave are sodic (Ahern, Rosenthal et al. 1988).

It was initially believed that deep infiltration from irrigation to the groundwater table would be less on the left bank due to the widespread distribution of heavy clay rich soils (QWRC 1985). The salinity and sodicity levels of these soils were later determined to be a potential problem for the region as a whole due to the amelioration that would be needed (Ahern, Weinand et al. 1989). While the technology exists to leach salts to lower depth or out of the profile and or modify sodicity with assistance of soil amendments such as gypsum, to produce crops at the individual farm level, excessive leaching and water use can result in salty and rising water tables (Ahern, Weinand et al. 1989). Subsurface drainage can be used to manage these problems in irrigation areas (Sharma and Gupta 2006) however this has not been implemented in the Lower Burdekin.

In the early stages of irrigation development a study into management options for irrigation of sodic soils in the area recommended the following strategies: ensuring good surface water drainage and planting on ridges to minimize surface water logging; improvement of soil structure prior to cropping and using rice crops to leach exchangeable sodium and improve soil structure; and where economically feasible, application of gypsum in combination with deep ripping (Smith and McShane 1981). Gypsum and lime are currently being applied to soils in the BHWSS to ameliorate soil sodicity. Gypsum sales from the “Burdekin Lime Company” in 1994 indicated that approximately 1100 hectares (10% of the BHWSS) were treated with gypsum during 1994 and that gypsum application rates were increasing (McMahon, Ham et al. 1996). Results from field trials have shown that gypsum applications in the Lower Burdekin can improve sugarcane yields (Ham, Cox et al. 1997), maintain soil macroporosity and decrease root zone salinity (Dowling and Elliot 1988) and improve the structure of surface soil (by increasing the concentration of electrolyte in the soil solution and helping to prevent dispersed clays from moving downwards in the soil profile) (Smith and McShane 1981).
However, the use of gypsum also increases water infiltration potentially leading to increases in deep drainage and groundwater recharge (Dowling and Elliot 1988; Dowling, Thorburn et al. 1991; PPK 2002).

The mixing of water from different sources with differing water quality characteristics is frequently used by Burdekin farmers but established guidelines for management are absent (Davis 2006). The use of high salinity blended irrigation waters (>0.8 dS/m) can lead to increased deep drainage characteristics and increased recharge on some soils (PPK 2002). Utilisation of irrigation water with a quality inappropriate to soil type properties can cause an array of productivity and environmental problems such as sodicity, salt accumulation in root zones, minimal water use efficiency or excessive deep drainage (Davis 2006). However, there is a distinct lack of measured data on the impacts of these practices.
Figure 2-5. Map showing the location of detailed soils mapping as the pale blue shapes with grey outlines defining the boundaries.
Figure 2-6. Detailed map of soils in the Mulgrave section of the BHWSS.
2.2.3 Groundwater Hydrology

Beneath the BHWSS, groundwater is present in the unconsolidated sediments. The hydrogeology of these aquifers is fairly complex due to the existence of paleochannels and the layering of clayey and sandy materials. Figure 2-7 shows an example of the hydrogeology in the Mulgrave area. The stratigraphic profiles are typically dominated by clays and sandy clays. The upper clay layer thickness can exceed ten metres and extend all the way to the underlying bedrock.

Since irrigation development commenced in the BHWSS, there have been several areas where rising groundwaters have been noted, including Mona Park and Mulgrave. The occurrence of rising groundwater in the BHWSS is illustrated in Figure 2-8. These maps were created in ArcMap using the elevation of measured groundwater levels in the wet season, January to February, in 1990 and 2006. The water table elevations are presented using Australian Height Datum (AHD). The location of the bores where the water levels were measured is included in the maps. The groundwater levels in the Mulgrave area south of Clare have risen by up to 10 metres during this period. Prior to irrigation development, the groundwater levels were generally 10 to 15 metres below ground level (Evans 1987).
Figure 2-8. Groundwater levels in the BHWSS in 1990 and 2006.
Prior to development, groundwater quality in the Mulgrave area was already quite saline with electrical conductivities in the alluvial material ranging from 2560 to 11000 µS/cm and electrical conductivities in the bedrock ranging from 1287 to 11400 µS/cm (Evans 1987). Deep drainage under irrigated fields can contribute additional salts to the underlying aquifers (Schoups, Hopmans et al. 2005). Extensive areas in the BHWSS scheme are already considered under threat of irrigation salinity including Mona Park, Giru and Leichhardt Plains (Roth, Lawson et al. 2002).

### 2.2.4 Knowledge Gap

A large proportion of the soils in the Mulgrave area of the BHWSS are sodic. These soils have been ameliorated to enable irrigation development, primarily by applying gypsum. Groundwater tables have been rising in this area and it is possible that widespread amelioration of the sodic soils could be contributing to rising groundwater tables by increasing the hydraulic conductivity beyond the root zone and into the unsaturated zone and thereby increasing potential deep drainage rates.

There is anecdotal evidence that gypsum applications have increased the infiltration rates of irrigation water but the magnitude and depth of changes in soil hydraulic properties is not known. In order to determine the potential for sodic soil amelioration to contribute to rising groundwater tables, the impacts of amelioration on the hydraulic properties of the local sodic soils needed to be measured. To do this, locations first needed to be selected for sampling of the local sodic soils.

### 2.3 Materials and Methods

The two key factors in selecting sites for soil sampling were the soil classification and the soil sodicity. The dominant soils in the BHWSS soils are cracking clays and duplex soils (Donnollan 1991) therefore soils maps from previous soil surveys were used to locate these soil types. Soil data from previous soil analyses (Ahern and Weinand 1989; Ahern, Weinand et al. 1989) was then used to restrict potential sampling sites to those that were
likely to be sodic. As the “Mulgrave” area contains the greatest proportion of sodic soils, this area was chosen for soil sampling.

Other factors that were considered were site accessibility and the availability of additional monitoring data at the site. Site accessibility was particularly important in cases where potential sites were outside of irrigated properties. Obstacles on these sites that precluded soil sampling included channels, power lines and vegetation. Access for sampling on irrigated properties required permission from the property owner. Field sites that were being used for other research projects were favoured if the soil types at these sites suited the objectives of this study.

The sites that were selected were in the northern part of the Mulgrave section. The soil types at the chosen sites were cracking clay and duplex soils (Figure 2-9). Sites 1, 3 and 5 were used to sample cracking clay soils. Sites 2 and 4 were intended for sampling duplex soils, however site 2 was close to the soil map boundary of duplex soils and cracking clay soils. Four of the sites selected, sites 1 to 4, are irrigated fields where sugarcane was growing. Two of these four sites, sites 3 and 4, were close to sites used for soil sampling prior to irrigation development. Site 5 is a native vegetation site adjacent to one of the irrigated sugarcane field sites.

Site 1 was found to be the most favourable site due to the combination of site accessibility and the presence of onsite monitoring as part of other research projects. As a result, the majority of the soil sampling and field measurements were carried out at this site.
2.3.1 Soil Sampling and Analyses

The purpose of the soil sampling was to collect samples for chemical and physical analyses, so that the soil could be characterised and the current sodicity levels could be determined, and to collect soil for use in laboratory column experiments. Soil samples were collected from multiple depths from sites 1 to 4 using a trailer mounted hydraulic soil sampling rig. At sites 1 and 2, three sampling points were selected at approximately 10 metre intervals along a transect. Soil sampling locations at these sites were limited by the size of the sugarcane crops. As a result, the samples had to be collected from the end of the rows of sugarcane.

The standard Queensland Government soil survey sampling depths were used, 0.0-0.1 m, 0.2-0.3 m, 0.5-0.6 m, 0.8-0.9 m, 1.1-1.2 m and 1.4-1.5 m. These depths were used to allow comparisons with previous studies. At site 2, samples were collected from additional depths further down the soil profile, below previous sample depths, to determine how the
Soil properties varied deeper in the profile. The extra sample depths were 1.6-1.7 m, 1.7-1.8 m, 2.0-2.1 m and 2.3-2.4 m.

In order to collect much larger samples from a specific horizon in the soil profile for use in the column experiments, a backhoe was used to dig trenches which provided access for soil sampling. The sites used in this case were site 1, an irrigated cracking clay site, and site 5, a nearby cracking clay site that had not been cleared for irrigation development. The trenches were dug using a 2 foot backhoe bucket and they were dug to a depth of 1.5 m. The soil profile was then studied to determine the depth interval that corresponds to the B horizon for the soil type of interest. The presence of calcareous nodules was used as an indicator for the horizon of interest. Bulk soil samples were then obtained for this depth range by using spades and sharp tools to dig soil from the wall of the trench. At site 5, soil samples were also collected at the depth intervals used for sites 1 to 4, namely 0.0-0.1 m, 0.2-0.3 m, 0.5-0.6 m, 0.8-0.9 m, 1.1-1.2 m and 1.4-1.5 m.

Soil samples were submitted to the Natural Resources Sciences Chemical Laboratories at Indooroopilly in Brisbane, a NATA accredited laboratory. The analyses performed included physical and chemical analyses. The measurement uncertainties and the lowest concentration of an analyte that can be reliably measured (the practical quantitation levels) for these analyses are provided in Table 2-1. The pH and EC of the soils were determined using 1:5 soil/water extracts following the method of Rayment and Higginson (1992). Particle size analysis was performed using the hydrometer method (Gee and Bauder 1986).

The method used for determination of exchangeable cations involved a pre-treatment to remove soluble salts and suppression of carbonate dissolution. Sixty per cent alcohol was used initially to remove soluble salts prior to extraction of cations by leaching with alcoholic 1 M ammonium chloride (NH₄Cl) at pH 8.5 at a soil to solution ratio of 1:20 (Tucker 1971). Exchangeable cation concentrations (calcium, magnesium, sodium and potassium) were then determined in the leachate using atomic absorption spectrometry.

The final stage, cation exchange capacity, was then determined by colorimetric finish using a continuous flow analyser, after displacing ammonium (and chloride) with a solution of 15% potassium nitrate (KNO₃) plus 6% calcium nitrate, 4-hydrate (Ca(NO₃)₂.4H₂O). This
K-Ca solution combination has a multiple capacity for effectively displacing ammonium from exchange sites (Loveday 1974).

Table 2-1. Measurement uncertainty and practical quantitation levels for the soil analyses, provided by the Natural Resources Sciences Chemical Laboratory, with the laboratory results.

<table>
<thead>
<tr>
<th>Analysis</th>
<th>Uncertainty %</th>
<th>Practical Quantitation Level</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electrical conductivity</td>
<td>10</td>
<td>0.01 dS/m</td>
</tr>
<tr>
<td>pH</td>
<td>5</td>
<td>0.1 pH units</td>
</tr>
<tr>
<td>Exchangeable Calcium</td>
<td>10</td>
<td>0.18 meq/100g</td>
</tr>
<tr>
<td>Exchangeable Potassium</td>
<td>12</td>
<td>0.015 meq/100g</td>
</tr>
<tr>
<td>Exchangeable Magnesium</td>
<td>8</td>
<td>0.31 meq/100g</td>
</tr>
<tr>
<td>Sodium</td>
<td>10</td>
<td>0.091 meq/100g</td>
</tr>
<tr>
<td>Exchangeable sodium percentage</td>
<td>0</td>
<td>0.1 %</td>
</tr>
<tr>
<td>Cation exchange capacity</td>
<td>10</td>
<td>3 meq/100g</td>
</tr>
<tr>
<td>Clay</td>
<td>5</td>
<td>1 %</td>
</tr>
<tr>
<td>Coarse sand</td>
<td>10</td>
<td>1 %</td>
</tr>
<tr>
<td>Fine sand</td>
<td>8</td>
<td>1 %</td>
</tr>
<tr>
<td>Silt</td>
<td>8</td>
<td>1 %</td>
</tr>
</tbody>
</table>

2.4 Soil Analysis Results

In this section, the results of the soil analyses are presented for all five sampling sites. The results that will be presented include the exchangeable sodium percentage, clay content and pH results at different depths. For sites 1, 3 and 4, comparisons are made with soil properties measured prior to irrigation development.

In general, all sites showed a trend of increasing sodicity with depth. Between 0.5 and 0.9 metres, the soils were typically in the “sodic” category according to Northcote and Skene (1972). Below 0.9 metres, some of the sites had ESPs much greater than 15, placing the lower parts of these soil profiles in the “strongly sodic” category (Northcote and Skene 1972). The soils mostly had pHs ranging from 7 to 9 which is typical of the sodic soils in this area.
At site 1, the exchangeable sodium percentages increased down the profile (Figure 2-10). Between 0 and 0.1 metres below the surface, the ESPs were below 6 and would be classified as non-sodic according to Northcote and Skene (1972). Between 0.2 and 0.8 metres, the ESPs had increased to between 6 and 12, in the “sodic” range. Below 0.8 metres, the ESPs were above 14, in the “highly sodic” range. The highest ESP measured was 25, at the deepest depth sampled, 1.5-1.6 metres. The variability between the replicates was the greatest at this depth.

In the soil surveys prior to development of the Mulgrave area, the chemical properties were measured for the different soil types (McClurg, Tucker et al. 1988). For comparison purposes, the results from these surveys for the soil type at site 1 are included (Figure 2-10). Although the ESPs were slightly higher for the “McClurg” soils when compared with the site 1 soils, the patterns for the ESP profiles for the “McClurg” soils are very similar to the soils sampled at site 1. The ranges of ESP values for each are also comparable. The soils at site 1 can therefore be assumed to provide a reasonable representation of the local cracking clay soils based on the sodicity levels.

Cracking clay soils (Ug) have a fairly uniform soil texture down the soil profile by definition (Northcote 1979). This can be seen in the profile for the “McClurg” soils in Figure 2-11. The clay percentages at site 1 showed more variation down the soil profile than was expected based on the McClurg, Tucker et al. (1988) results (Figure 2-11). The

Figure 2-10. Exchangeable Sodium Percentages at Site 1.
majority of the samples fell into the 50-60% clay range like those for McClurg, Tucker et al. (1988). Replicates 2 and 3 displayed a maximum clay percentage between 0.5 and 1 metres below the surface, followed by a decrease in clay percentage. Replicate 1 showed an unusual trend where the clay percentage decreased significantly at around 0.5 metres before increasing again. It is possible that mixing of the natural soil profile has occurred when the soil was prepared for cropping.

![Figure 2-11. Clay Percentages at Site 1.](image)

The general trend for the pH results at site 1 is an increase from between 7 and 8 near the surface to between 8 and 9 further down the profile (Figure 2-12). This trend is similar to the results from McClurg, Tucker et al (1988) Figure 2-12. The exceptions to this general trend for the data from site 1 are mostly found in replicate 1 where the pH values are below 7 in the top 60 cm of the profile.
It was determined after sampling at site 2 that the soil type appeared to be on the borderline between two mapped soil types namely, a cracking clay and a sodic duplex soil. The clay percentages at this site 2 are lower than those at site 1 but the clay percentages down the profile do not follow the expected trend for duplex soils of a higher clay content in the B horizon compared with the A horizon. The variability between replicates can be seen most clearly in the clay percentage results (Figure 2-13). While replicates 1 and 2 were generally in the 50 to 60% range for the top 1 metre of the soil profile, the third replicate had a much lower clay content, between 35 and 45% in the top 1 metre. The ESP levels were generally strongly sodic below the 0.5 metre depth (Figure 2-14). The pH values were fairly uniform down the profile and typically ranged from 8 to 9.5 (Figure 2-15).

At site 2, deeper sampling was undertaken to determine whether high sodicity levels and high clay contents were present below the traditional sampling depths. At this site, there was a decline in the clay percentage below 1.5 metres. This indicates a shift from a “clay” soil to a “clay loam” soil. Sodicity levels were found to increase down the profile with ESP levels above 25 at depths greater than 1.5 metres. It is not known if this same trend occurs at other locations as sampling typically does not extend below this depth.
Figure 2-13. Clay percentages at site 2.

Figure 2-14. Exchangeable Sodium Percentages at site 2.
Sites 3 and 4 were adjacent to sites sampled previously as part of the pre-development soil surveys. These soil surveys were undertaken in 1978. Comparisons have been made between the soil properties measured in 1978 and the soil properties measured for this study in 2007. It was expected that the sodicity and salinity levels of the upper soil profiles would have been reduced since 1978 due to the effects of leaching with irrigation water.

At both sites, the EC appears to have decreased between 1978 and 2007 (see Table 2-2 and Table 2-3). The concentrations of exchangeable sodium have also decreased but at most depths, this reduction was in the order of less than 3 meq/100g. At the same time, exchangeable magnesium concentrations generally decreased slightly and exchangeable calcium concentrations did not show a clear trend between the two sampling periods. However, as no replicates were used, the standard deviation for these measurements is not known and it is not possible to determine whether there are statistically significant differences between the two sets of results.
### Table 2-2. Soil properties at site 3.

<table>
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<tr>
<th>Depth (m)</th>
<th>EC (dS/m)</th>
<th>Ca (meq/100g)</th>
<th>Mg (meq/100g)</th>
<th>Na (meq/100g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.00-0.10</td>
<td>0.13</td>
<td>0.05</td>
<td>-0.08</td>
<td>25</td>
</tr>
<tr>
<td></td>
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<td></td>
</tr>
<tr>
<td>0.20-0.30</td>
<td>0.15</td>
<td>0.08</td>
<td>-0.07</td>
<td>21</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>0.50-0.60</td>
<td>0.30</td>
<td>0.20</td>
<td>-0.10</td>
<td>16</td>
</tr>
<tr>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>0.80-0.90</td>
<td>0.55</td>
<td>0.46</td>
<td>-0.09</td>
<td>13</td>
</tr>
<tr>
<td></td>
<td></td>
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<td></td>
</tr>
<tr>
<td>1.10-1.20</td>
<td>0.91</td>
<td>0.72</td>
<td>-0.19</td>
<td>12</td>
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<td></td>
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<td></td>
</tr>
<tr>
<td>1.40-1.50</td>
<td>1.00</td>
<td>0.77</td>
<td>-0.23</td>
<td>10</td>
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<tr>
<td></td>
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</tr>
</tbody>
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### Table 2-3. Soil properties at site 4.

<table>
<thead>
<tr>
<th>Depth (m)</th>
<th>EC (dS/m)</th>
<th>Ca (meq/100g)</th>
<th>Mg (meq/100g)</th>
<th>Na (meq/100g)</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.00-0.10</td>
<td>0.04</td>
<td>0.06</td>
<td>+0.02</td>
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<td>0.20-0.30</td>
<td>0.38</td>
<td>0.22</td>
<td>-0.16</td>
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</tr>
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<td>0.50-0.60</td>
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<td>0.60</td>
<td>-0.70</td>
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<tr>
<td>1.10-1.20</td>
<td>1.20</td>
<td>0.39</td>
<td>-0.81</td>
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</tr>
<tr>
<td>1.40-1.50</td>
<td>0.65</td>
<td>0.40</td>
<td>-0.25</td>
<td>5.6</td>
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</tbody>
</table>

Site 5, the only site with native vegetation rather than irrigated sugarcane, was used for comparison with the irrigated cracking clay soils. It was expected that ESP and EC levels would be lower at the irrigated site in response to application of good quality irrigation water and subsequent leaching of soil salts. However, there are a number of other factors that would be expected to contribute to differences in soil properties between the two sites. These factors include soil variability within a mapping unit and temporal variability in soil properties. Ideally many more sampling sites would be needed to make comparisons.
between irrigated sites and natural sites. These limitations need to be kept in mind for the comparisons made between the two sites in this chapter.

The results presented here show comparisons between site 1 and site 5. The average of the three replicates is shown for each site with the errors bars showing 1 standard deviation either side of the average in Figure 2-16, Figure 2-17 and Figure 2-18.

Qualitatively, there appears to be different trends in clay percentages. At site 5 the clay percentages are above 60% in the top metre of soil and less than 50% below this (Figure 2-16). Site 1 exhibited a more uniform soil texture of between 50% and 60% down the profile. Mixing of the soil profile is a possible cause for the more uniform texture at site 1 when compared to site 5. However, there were only statistically significant differences in clay percentage between the two sites, at a probability level of 0.05, at the top and bottom of the profile, at 0 – 10 cm, 110 – 120 cm and 140 – 150 cm respectively.

![Figure 2-16](image)

Figure 2-16. Comparison of clay percentages at sites 1 and 5, where error bars show 1 standard deviation either side of the mean and asterices indicate significant differences between the sites at a 0.05 probability level.

At both site 1 and site 5 the ESP increased with depth (Figure 2-17). Overall, there was no significant difference between the ESPs at site 1 and site 5 at a probability level of 0.05 with the exception of the ESPs measured at 50-60 cm. The similarity in sodicity levels is despite the effects of leaching with irrigation water. This may indicate that leaching with
irrigation water has had minimal impacts on soil sodicity. Alternatively, the sodicity level prior to irrigation development at site 1 may have been much higher than the current sodicity at site 5.

Figure 2-17. Comparison of ESPs at sites 1 and 5, where error bars show 1 standard deviation either side of the mean and asterices indicate significant differences between the sites at a 0.05 probability level.

The electrical conductivity profiles at site 5 were significantly different, at a probability level of 0.05, from those at site 1 at most depths, namely 0.2-0.3 metres, 0.8-0.9 metres, 1.1-1.2 metres and 1.4-1.5 metres. The main difference between the two sites is the increase in EC between 0.8-0.9 and 1.1-1.2 metres at site 5 (Figure 2-18). At site 1 the trend down the profile shows a steady increase below 0.2-0.3 metres.

There are also differences in the EC levels near top of the soil profile. The EC at site 1 at 0.2-0.3 metres was higher than the EC at site 5 at 0.2-0.3 metres. It is possible that the higher salt content at this point is the result of a concentration of salts in response to evapotranspiration. Alternatively the differences may be due to the salt levels originally being higher at this location.
X-ray Diffraction (XRD) analyses were performed to determine:

a) whether these soils contained “swelling clays” (smectites)
b) the approximate percentage of smectites and
c) the dominant clay minerals to allow comparison of these soils to other soils studied in the literature.

Qualitative XRD analyses were performed at the University of Queensland’s Centre for Microscopy and Microanalysis for soil samples from 3 depths at site 1 and 5 depths at site 2. The clay mineral types detected in all of the samples were kaolinite, smectite and illite. Figure 2-19 is a representative XRD pattern for the qualitative results. The green lines, dark blue lines and red lines show the expected positions for smectites, illites and kaolinites respectively. The presence of smectites was confirmed by analysing glycolated samples.
Quantitative XRD analyses were then performed at the Queensland University of Technology’s X-ray Analysis Facility. As the same clays were found to be present at all depths at both sites when the qualitative analyses were performed, quantitative analyses were not performed for every site. Instead, only samples from the same site and depth interval as the soil used in laboratory experiments were used. The results are shown in Table 2-4. These percentages were calculated based on the detected amounts of illite, kaolinite and smectite. Any clay minerals present that could not be identified, for example mixed layer clay types, would introduce errors to these calculated percentages.

Table 2-4. Quantitative XRD results for samples from the B horizon of site 1.

<table>
<thead>
<tr>
<th></th>
<th>Illite</th>
<th>Kaolinite</th>
<th>Smectite</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average</td>
<td>19.3%</td>
<td>69.6%</td>
<td>9.9%</td>
</tr>
<tr>
<td>Standard Deviation</td>
<td>3.2%</td>
<td>4.0%</td>
<td>1.5%</td>
</tr>
</tbody>
</table>
2.5 Discussion

The dominant soil types in the BHWSS are cracking clays and duplex soils. These soil types were targeted as being typical of the region. It is, however, understood that within these soil types there is likely to be significant variation in soil properties. As the number of sampling locations in this study was limited to five sites in the northern part of the “Mulgrave” area, it is unlikely that the full variability of soil properties within these soil types in the BHWSS was captured.

The soil sampling for this study was also biased towards soils that were likely to be sodic so that the impacts of amelioration on sodic soils could be studied. Although the irrigated soils were expected to exhibit reduced sodicity levels, the results of the analyses for the sites used in this study did not support this. There are a number of reasons why this may be the case. Firstly, there could be differences between the exact sites sampled prior to development and those sampled in 2007. At the sites where the soil profile was sampled at 10 metre intervals along a transect, the soil properties were sometimes found to vary significantly over these relatively small distances. Secondly, the full history of irrigation and soil treatments was not known at these sites. The impacts of irrigation water on soil sodicity depend on the quantity and quality of irrigation water. While saline irrigation waters are expected to increase infiltration rates due to electrolyte effects, only irrigation waters with low sodium adsorption ratios are expected to decrease soil sodicity.

In the case where comparisons were made between an irrigated site and a native vegetation site, sites 1 and 5 respectively, the sodicity levels were similar. The comparison between these two sites was based on a paired sites approach, where two adjacent sites are chosen with the same soil type and different land use types (Radford, Silburn et al. 2009). There were differences between sites 1 and 5 in the depth of the maximum salt concentration which may indicate downward movement of salts due to increased leaching with irrigation water at site 1. However, deeper sampling would be needed to assess changes in salinity profiles (Radford, Silburn et al. 2009). In addition, there were quite large differences in the clay percentages measured at different depths in the profile at these two sites. These differences in soil texture may indicate slight differences in the specific soil type, even though both soils were classified as cracking clays.
In general, the ranges of measured values and the trends down the soil profile for clay percentage, ESP and pH were similar to those reported prior to development (McClurg, Tucker et al. 1988). This was particularly the case for the cracking clay (Ug) soils. At site 2, there was significant variation between replicates, possibly due to spatial variations in soil properties at this location. At this site, sampling to depths of 2.5 metres showed that the ESP levels were still high below the root zone. This finding is important because in earlier soil surveys, sampling depths only extended to 1.5 metres. Information about sodicity levels below 1.5 metres was previously not available.

The presence of deep profiles of sodic clays may slow down the impacts of sodic soil amelioration. The results from site 2 indicate that these sodic clays extend to depths of at least 2.5 metres. Bore statigraphy logs have shown that clay layers in the BHWSS can extend to depths of up to 10 metres (Figure 2-7). In this study, sodicity was not measured below 2.5 metres but given the high levels at this depth it is likely that the clay soils below this point would also be sodic.

While there was a continuous increase in ESP down the profile, the pH values remained fairly constant, even decreasing slightly lower in the profile. This differs from the results of Nelson, Lawer et al. (2002) who found significant positive correlations between pH and ESP for Burdekin soils. As pH is sometimes used as an indicator for sodic soils in the Burdekin, the relationship between these properties is considered to be important. From the results in this study, it appears that the positive correlation between pH and ESP does not occur for all sodic soils in the Lower Burdekin. Day, Loi et al. (1992) stated that the correlation between pH and ESP was strongest for sodosols.

The majority of the soil sampling was conducted at site 1. Soil from this site was also used for the column experiments described in Chapter 3. The results of soil sampling and analyses at site 1 showed that the soil properties were fairly typical of the properties of cracking clays in the Mulgrave area of the BHWSS. This soil type is also the dominant soil type in the BHWSS. The soil collected from site 1 was therefore assumed to be representative of the cracking clay soils in the BHWSS.
Clay mineralogy has previously been shown to influence of chemistry on soil hydraulic properties (Frenkel, Goertzen et al. 1978). The clay soils in the BHWSS were found to have a mixed mineralogy, containing kaolinite, illite and smectite. Clays rich in smectite have previously been found to be more responsive while clays rich in kaolinite were found to be virtually insensitive to solution chemistry (McNeal and Coleman 1966). However, other results have shown that for cracking clays soils, soils with high clay contents and smectite mineralogy were found to be less susceptible to decreases in hydraulic conductivity at a given sodicity than soils of moderate clay content and mixed mineralogy (Shaw, Brebber et al. 1994). So the significance of clay mineralogy in controlling the impacts on the relationship between soil chemistry and hydraulic conductivity appears to be modified by other soil properties like clay content.

2.6 Conclusions

Groundwater levels have been rising beneath the BHWSS. It is possible that widespread amelioration of sodic soils could be contributing to these rising water tables. Duplex soils and cracking clays both exhibit high sodicity levels and these are the dominant soil types in the BHWWS. Within the BHWSS, the Mulgrave area has the greatest area of sodic soils.

A number of sites were selected in the Mulgrave area for soil sampling and analyses. The soil types at most of these sites were cracking clays. According to the definitions of Northcote and Skene (1972), all of the sites sampled were “sodic” below 0.5 metres and at some of the sites, the soils were “strongly sodic” deeper in the profile. High sodicity levels were even detected below the root zone, to depths of 2.5 metres. These high sodicity levels were present even at sites that had been irrigated for a number of years.

The sampled soils generally had high clay contents and the clay percentage was often fairly constant down the profile. Clay contents were found to be high even below 1.5 metres. Across all sites and depths, the clay contents were typically greater than 40%, with a maximum clay content of 67% measured at site 5.
Soil from site 1 was collected for use in laboratory column experiments. Although there are limitations with using soil from only one site for the column experiments, the soil properties at this site were found to closely resemble a “typical” cracking clay soil for this area based on the results of McClurg, Tucker et al. (1988) and cracking clays are the dominant soil type in the BHWSS.

The high clay content and high sodicity levels for the soils sampled in this study might be expected to slow down the impacts of sodic soil amelioration and limit the depth to which the soil properties would be altered due to the likelihood of low hydraulic conductivities. As the soil has a mixed mineralogy, the expected response to changes in soil chemistry as a result of clay mineral type from the literature can not be applied directly to predict the response of the soils from the BHWSS. Therefore, laboratory experiments and modelling are required to determine how these soils respond to amelioration treatments.
CHAPTER 3 – IMPACTS OF APPLYING GYPSUM
AND DIFFERENT SALINITY IRRIGATION
WATERS ON THE HYDRAULIC CONDUCTIVITY
OF SODIC CLAY SOILS

3.1 Introduction

In the BHWSS, large areas of the soils are sodic. These soils have been treated in order to improve infiltration of irrigation water and the plant available water in the root zone. Treatment methods have included applying gypsum and increasing the salinity of irrigation water by adding saline groundwater to the river water used for irrigation. While there is anecdotal evidence of infiltration rates being increased, the magnitude of the impacts of these treatments on local soil hydraulic conductivity in response to changes in soil chemistry has not been quantified.

This Chapter presents the results of column leaching experiments where gypsum solutions or mixed cation solutions of varying salinity were applied to sodic clays from the BHWSS and the hydraulic conductivity was measured. Changes in leachate chemistry during column leaching are studied to help explain the concurrent changes in hydraulic conductivity.

These experiments focus on finding out how the soils from the BHWSS respond to amelioration treatments. However, it is recognised that the laboratory conditions differ from those in the field and as such there are some limitations in using laboratory experiments to study the impacts of amelioration treatments. In particular, the continuous application of different treatments is not expected to occur in the field but is used in the laboratory experiments to determine the potential impacts of long term treatments in the field. In addition, repacked soils are used for the laboratory experiments and this is likely
to lead to differences between the measured hydraulic conductivity in the laboratory and actual hydraulic conductivity in the field.

A review of available literature is summarised in this Chapter with the aim of outlining the established relationships between soil chemistry and soil hydraulic properties as well as assessing experimental methods that have been applied for similar studies. The experimental methodology used in this study is then described before the results are presented. The results presented in this Chapter show changes in hydraulic conductivity as a function of leaching time and pore volumes; the changes in soil chemistry which are associated with changes in hydraulic conductivity; and the influence of bulk density of the packed columns on hydraulic conductivity.

### 3.2 Literature Review

The permeability of sodic soils is related directly to the pore-size distribution, which in turn depends on the texture of the soil and the extent of swelling of the clay fraction (Gardner, Mayhugh et al. 1959), with the swelling being controlled by soil sodicity and the salinity of applied water. The exchangeable sodium ions on clay surfaces decrease soil permeability by increasing clay swelling and modifying pore structure. Increasing the salinity of the irrigation water applied causes flocculation of clay particles such that the permeability of soils with high sodicity levels can be maintained if a sufficiently strong electrolyte solution is used (Quirk and Schofield 1955).

Changes in soil permeability in response to changes in solution composition can be predicted qualitatively based on clay mineralogy (McNeal and Coleman 1966). Despite many laboratory studies (Aringhieri and Giachetti 2001; Aydemir and Najjar 2005; Bagarello, Iovino et al. 2006; Barlow and Nash 2002; Cass and Sumner 1982; Chaudhari and Somawanshi 2004; Dane and Klute 1977; Eltaif and Gharaibeh 2007; Frenkel, Goertzen et al. 1978; Frenkel, Levy et al. 1992; Jayawardane 1979; Keren and Singer 1988; Levy, Goldstein et al. 2005; Loveday 1976; Mace and Amrhein 2001; McNeal and Coleman 1966; Melkamu, Yano et al. 1997; Pupisky and Shainberg 1979; Quirk and
Schofield 1955; Rhoades and Ingvalson 1969; Russo and Bresler 1977b; Shainberg and Caiserman 1971; Shainberg, Levy et al. 2001; Shainberg, Rhoades et al. 1981a; Suarez, Rhoades et al. 1984; Suriadi, Murray et al. 2002), it is not yet possible to accurately predict changes in the hydraulic conductivity of different soils based on the composition of the applied solution alone (McNeal and Coleman 1966; Russo and Bresler 1977b). This is partly due to differences between the specific soils, or extracted clays, tested and the soils of interest. Laboratory studies have also highlighted other important factors in controlling soil responses to sodic and saline conditions, including clay mineral dissolution (Shainberg, Rhoades et al. 1981b), the presence of oxides (Alperovitch, Shainberg et al. 1985), the presence of various organic and inorganic anions (Frenkel, Levy et al. 1992) and pH (Suarez, Rhoades et al. 1984). More recent studies have begun to recognise the importance of additional factors which are not related to properties of the soil or applied solution. These include time dependent experimental factors related to wetting of the soil columns such as, prewetting rates before continuous leaching (Shainberg, Levy et al. 2001), wetting rates (Levy, Goldstein et al. 2005) and cyclic wetting and drying cycles (Guney, Sari et al. 2007). These factors impact soil structural stability due to changes at the aggregate, or particle, scale.

3.2.1 Measurements of the Impact of Solution Chemistry on Soil Hydraulic Conductivity

To measure the impacts of soil chemistry on hydraulic conductivity it is necessary to leach the column with various compositions of applied solution and measure the responding changes in hydraulic conductivity. The constant head method (Klute and Dirksen 1986) is commonly used and the hydraulic conductivity is determined by measuring the drainage rate (Frenkel, Goertzen et al. 1978).

The compositions of the applied solutions varied widely between different published studies, from solutions saturated with gypsum (CaSO₄) to mixed cation solutions. There have been a significant number of experiments published that have leached soil columns with mixed cation salt solutions with different sodium adsorption ratios (SARs) (Aringhieri and Giachetti 2001; Bagarello, Iovino et al. 2006; Cass and Sumner 1982; Chaudhari and
Chapter 3 Impacts of applying gypsum and different salinity waters on hydraulic conductivity

Somawanshi 2004; Dane and Klute 1977; Frenkel, Goertzen et al. 1978; Jayawardane 1992; Keren and Singer 1988; Mace and Amrhein 2001; Melkamu, Yano et al. 1997; Pupisky and Shainberg 1979; Russo and Bresler 1977a; Shainberg and Caiserman 1971). The mixed cation solutions used are commonly composed of sodium chloride and calcium chloride (Aringhieri and Giachetti 2001; Dane and Klute 1977; Pupisky and Shainberg 1979). The purpose of using mixed cation solutions with varied concentrations of sodium and varied total salt concentrations is to determine the effects of sodicity and salinity on the hydraulic conductivity of soils of interest. This has implications for determining whether a given water quality is suitable for irrigation, particularly where irrigation with wastewater is being considered (Bhardwaj, Mandal et al. 2008).

There are some variations in the methods used to determine the influence of sodicity and salinity on hydraulic conductivity. In some cases, the soil is first equilibrated with a solution of fixed SAR before leaching with deionised water or low salinity water and measuring the changes in hydraulic conductivity (Keren and Singer 1988; Melkamu, Yano et al. 1997). These studies are used to determine the stability of sodic soils when low salinity waters are applied. But in most cases, the soil columns were leached with a series of solutions with fixed SARs and decreasing salt concentrations, with flow rate determined for each combination of SAR and salt concentration (Aringhieri and Giachetti 2001; Cass and Sumner 1982; Dane and Klute 1977; Pupisky and Shainberg 1979). These experiments follow on from the work of McNeal and Coleman (1966).

The results for the effect of solution composition on soil hydraulic conductivity by McNeal and Coleman (1966) have been widely cited. This series of experiments measured the decreases in hydraulic conductivity with decreases in electrolyte concentration and increasing sodium adsorption ratio (SAR) of the percolating solution for seven soils of varying clay mineralogy. The purpose of these experiments was to provide information on the effect of a wide range of salt solutions on the hydraulic conductivity of several well characterized soils. The effects of solution composition on hydraulic conductivity were determined by sequentially applying solutions with decreasing salt concentrations, from 800 meq/L to 3.13 meq/L, and constant SARs. The SARs used for these experiments were 0, 14, 25, 50, and 100. The results for two soils were presented as graphs of the measured
hydraulic conductivity for combinations of SAR and salt concentration and the results for the Pachappa sandy loam are shown here (Figure 3-1).

![Figure 3-1. The hydraulic conductivity vs. salt concentration for Pachappa sandy loam, with no detail shown within 5% of the line for SAR = 0 (McNeal and Coleman 1966).](image)

Following on from these experimental results, a model was developed to relate decreases in hydraulic conductivity to soil swelling associated with increasing soil SAR or decreasing salt concentration of the applied solution (McNeal 1968). The general equation developed provided a satisfactory match with the experiment results (McNeal 1968):

\[
1 - y = cx^n/(1 + cx^n)
\]  

(3.1)

where \(y\) is the relative soil hydraulic conductivity (the measured hydraulic conductivity divided by the optimum hydraulic conductivity), \(x\) is a swelling factor (the calculated interlayer swelling of soil montmorillonite) and \(c\) and \(n\) are constants for a given soil within a specified range of ESP values. As a first approximation, acceptable \(n\) values are \(n = 1\) for ESP < 25, \(n = 2\) for ESP > 25 & <50 and \(n = 3\) for ESP >50. Only values of \(c\) need to vary between different soils (McNeal 1968).

A method is suggested by McNeal (1968) to obtain \(c\) values for different soils: measure the relative hydraulic conductivity with a high salt, high SAR solution (e.g. salt concentration of 800 meq/L and SAR of 50 or 100). By then maintaining the same SAR while
decreasing the salt concentration of the percolating solution to a relatively low value (e.g. 50 meq/L), the relative hydraulic conductivity can be determined (McNeal 1968).

In the original experiments (McNeal and Coleman 1966), ten pore volumes of initial solution were used to equilibrate the soil and high salt concentration water. Approximately five pore volumes of each subsequent solution were then used for leaching before the applied solution was changed (Figure 3-2).

![Figure 3-2. The hydraulic conductivity of Pachappa sandy loam at SAR = 0, SAR = 25, and SAR = ∞, the numbers beside the small arrows indicate the salt concentration of the solution introduced at that point (McNeal and Coleman 1966).](image)

### 3.2.2 Measured Impacts of Gypsum Applications on Hydraulic Conductivity

The impacts of gypsum applications on the hydraulic conductivity have been measured in a number of laboratory experiments (Aydemir and Najjar 2005; Loveday 1976; Mace and Amrhein 2001; Scotter 1985; Shainberg, Keren et al. 1982), but the magnitude of these impacts are expected to be soil specific. The soils used in these experiments ranged from loams (Aydemir and Najjar 2005) to clays (Loveday 1976; Scotter 1985). While the clay mineralogy varied between the different soils tested, it was commonly dominated by montmorillonite (Aydemir and Najjar 2005; Prather, Goertzen et al. 1978; Scotter 1985).
This differs from the soils of interest in this study where the clay percentage was above 60% and the clay mineralogy was dominated by kaolinite.

While some of the studies used gypsum in solution (Scotter 1985), many of the studies involved mixing varying quantities of gypsum into the soil, often spreading it on top of the soil column (Aydemir and Najjar 2005; Mace and Amrhein 2001; Shainberg, Keren et al. 1982). When gypsum is mixed into the soil before leaching, the hydraulic conductivity begins to reduce once the impacts of the applied gypsum wear off (Aydemir and Najjar 2005; Mace and Amrhein 2001; Shainberg, Keren et al. 1982). In these cases, the maximum hydraulic conductivity was measured soon after leaching commenced. Even when excess gypsum is mixed into the soil before leaching, some of the exchangeable sodium can remain in the soil because a proportion of the applied calcium passes through the column (Mace and Amrhein 2001). For the study where a saturated gypsum solution was used (Scotter 1985), only one to two pore volumes of this solution were infiltrated through the soil and the hydraulic conductivity was still increasing when the applied solution was changed to a sodium chloride solution or distilled water.

In previous studies (Loveday 1976; Mace and Amrhein 2001), the impacts of gypsum were determined by measuring the changes in soil hydraulic conductivity within the first 48 hours after leaching was commenced. The use of single gypsum applications or short leaching times in previous studies did not enable the full impacts of gypsum on hydraulic conductivity to be measured (Mace and Amrhein 2001; Prather, Goertzen et al. 1978), leading to a potential underestimation of the impacts of gypsum on sodic soils.

It is difficult to predict the exact impact of gypsum applications on the hydraulic conductivity of individual sodic soils, despite having a reasonable understanding of the mechanisms involved, because there are number of factors involved. The amount of gypsum required to reclaim a sodic soil depends on exchangeable cation concentrations, cation exchange capacity, soil solution composition, bulk density and the depth of soil to be reclaimed (USDA 1954). In addition, the efficacy of gypsum treatments depends on the quality of the gypsum product (Nelson, Ham et al. 2001), the rate of gypsum dissolution (Keren and Shainberg 1981), the rate of leaching (Keren and O'Connor 1982) and the composition of the leaching solution (Hulugalle, Weaver et al. 2006; Mace and
Amrhein 2001). While the influence of some of these factors has been studied previously, the expected effects of specific combinations of these factors cannot be predicted without testing the soils of interest.

### 3.2.3 Experimental Considerations when Measuring the Effect of Solution Composition on Soil Hydraulic Conductivity

When measuring the impacts of solution chemistry on soil hydraulic properties, there are a number of factors that influence the results. Many of these factors can be controlled under laboratory conditions. These factors include hydraulic gradient, column size, bulk density of the soil in packed columns and separation of the clay fraction.

Constant head measurements of hydraulic conductivity are based on the application of Darcy’s Law. For Darcy’s Law to be valid there needs to be a linear relationship between flow rates and hydraulic gradients. Increasing the hydraulic gradient to greater than unity can lead to non-linear relationships between flow rates and hydraulic gradients, possibly as a result of flow restrictions caused by the progressive blocking of pores by migrating soil particles (Madsen, Chandler et al. 2008). It is common to use large hydraulic head gradients when testing low permeability soils to decrease measurement time (Madsen, Chandler et al. 2008). In previous experiments where the impacts of solution composition on hydraulic conductivity were measured, the hydraulic gradient was often greater than 1 (Aringhieri and Giachetti 2001; Cass and Sumner 1982; Jayawardane 1979; Rhoades and Ingvalson 1969).

Increases in bulk density cause increases in soil compaction that can result in substantial decreases in hydraulic conductivity (Assouline 2006). Bulk density has been found to alter measured hydraulic conductivities for kaolinitic soils of varying sodicity (Frenkel, Goertzen et al. 1978). However, the majority of previous experiments studying the effects of solution chemistry on hydraulic conductivity used only one bulk density (Frenkel, Levy et al. 1992; Jayawardane 1979; Pupisky and Shainberg 1979; Shainberg, Bresler et al. 1971; Shainberg and Caiserman 1971; Suarez, Rhoades et al. 1984; Suriadi, Murray et al. 2002).
To determine how the hydraulic conductivity of different clay types responds to changes in solution chemistry, experiments have been conducted using extracted clays (Aringhieri and Giachetti 2001; Frenkel, Levy et al. 1992; McNeal and Coleman 1966; Melkamu, Yano et al. 1997; Shainberg and Caiserman 1971). The clay mineralogy for the soils tested included illites, kaolinites, vermiculites, chlorites, gibbsite and montmorillonites. Natural soils often contain multiple types of clay minerals, and sometimes the clay minerals are even interlayered making it more difficult to base predictions on responses of individual extracted clays.

The hydraulic conductivity responses of different soils to changes in solution chemistry are primarily the result of clay mineralogy and soil type. Hydraulic conductivity has been found to decrease more drastically with increasing clay content (Frenkel, Goertzen et al. 1978). The soil textures that have been tested in previous studies include loams (Russo and Bresler 1977a) silty loams (Quirk and Schofield 1955), sandy loams (Gardner, Mayhugh et al. 1959; Pupisky and Shainberg 1979) and clay soils (Loveday 1976). The sodic clays tested by Loveday (1976) had a clay content of 59%, significantly higher than most other studies that used soils with clay contents below 50% (Dane and Klute 1977; Frenkel, Goertzen et al. 1978; Quirk and Schofield 1955).

In addition to measuring changes in hydraulic conductivity in response to changes in solution composition, measurement of the composition of the leachate solution can be used to determine what changes are occurring in the soil solution chemistry and when equilibration with the applied solution has occurred. The choice of which aspects of effluent solution chemistry should be measured depends on the composition of the applied solution. Measurement of multiple aspects of the leachate solution chemistry can be useful to confirm equilibration. Some of the aspects of leachate solution composition that have been measured previously include calcium concentrations (Dane and Klute 1977; Suarez, Rhoades et al. 1984), electrical conductivity (Aringhieri and Giachetti 2001; Frenkel, Goertzen et al. 1978; Levy, Goldstein et al. 2005; Pupisky and Shainberg 1979; Shainberg, Levy et al. 2001; Suriadi, Murray et al. 2002), pH (Frenkel, Goertzen et al. 1978; Keren and Singer 1988; Levy, Goldstein et al. 2005; Shainberg, Levy et al. 2001; Suriadi, Murray et al. 2002) and dispersed clay concentrations (Aringhieri and Giachetti 2001; Bagarello,
3.3 Materials and Methods

Two sets of experiments were conducted to determine the changes in soil hydraulic conductivity that occur when a) a saturated gypsum solution and b) mixed cation solutions of varying salinity are applied. The methods used are briefly introduced here and explained in detail in the following sections. The (bulk) soil used in these experiments was collected from site 1 (described in section 2.3). The soil was dried, ground, sieved to < 2 mm and repacked to the desired bulk density.

In the first set of experiments, a saturated gypsum solution was applied and the primary aim was to measure the increase in hydraulic conductivity when gypsum is applied. A low salinity irrigation water surrogate was used for comparison with the gypsum solution results.

Using the method described in section 3.3.2 for packing the columns, the maximum bulk density that could be achieved was 1.4 g/cm$^3$. As the measured field bulk densities (approximately 1.9 g/cm$^3$) were much higher than the bulk densities of the repacked columns, the influence of bulk density needed to be assessed so that the hydraulic conductivity results could be extrapolated with confidence to the field conditions. For this reason, the experiments with gypsum solutions applied were conducted with columns packed at both 1.3g/cm$^3$ and 1.4g/cm$^3$. Five replicates were used for each of these bulk densities.

The second set of experiments was designed to determine the influence of salt concentration on hydraulic conductivity, after the method of McNeal (1968). A bulk density of 1.3 g/cm$^3$ was used for these experiments as preliminary experiments had shown faster responses when a bulk density of 1.3 g/cm$^3$ was used. Two salt concentrations were
used for these experiments, with three replicates for each salt concentration. The two sets of experiments were carried out simultaneously.

### 3.3.1 Materials

The soil samples used in this study were collected from an irrigated sugarcane field in the Lower Burdekin in North Queensland. The focus of this study was to determine the potential changes in saturated soil hydraulic conductivity at the bottom of the root zone in response to long term gypsum applications. Samples were therefore collected from the B horizon, 0.6 to 1.1 metres below the ground surface. This horizon typically exhibits higher sodicity levels than the A horizon (see section 2.4). The soils tested are alkaline, clay soils with high sodicity levels (Table 3-1). The ESP of these soils places them in the strongly sodic category, i.e. ESP > 14, based on Australian soil classifications (Northcote and Skene 1972). The sodicity levels in the B horizon are typically higher than those in the A horizon but the clay contents are similar (section 2.4). The dominant clay minerals are illite, smectite and kaolinite (section 2.4).

Some intact cores were collected at site 1 from the 0.8-0.9 metre depth interval, using a sharp edged sampling device containing Perspex columns of 4 cm height and 5.4 cm internal diameter. Excess soil was removed from the ends of the intact cores by gently scraping the soil from the centre using a sharp serrated edged knife. Some of these intact cores were used in the laboratory experiments however an insufficient number of cores were collected to provide replicates for each of the treatments so repacked columns were also needed.
Table 3-1. Typical soil properties for the soils used in these experiments, based on five samples from the same site.

<table>
<thead>
<tr>
<th></th>
<th>pH</th>
<th>EC (dS/cm)</th>
<th>Exchangeable Cations (meq/100g)</th>
<th>ESP (%)</th>
<th>CEC (meq/100g)</th>
<th>Clay (%)</th>
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<tr>
<td></td>
<td></td>
<td></td>
<td>Ca</td>
<td>Mg</td>
<td>Na</td>
<td></td>
</tr>
<tr>
<td>A</td>
<td>8.8</td>
<td>0.51</td>
<td>11.1</td>
<td>14.3</td>
<td>5.2</td>
<td>17.3</td>
</tr>
<tr>
<td>B</td>
<td>8.8</td>
<td>0.46</td>
<td>10.2</td>
<td>13.6</td>
<td>4.9</td>
<td>16.9</td>
</tr>
<tr>
<td>C</td>
<td>8.8</td>
<td>0.48</td>
<td>10.9</td>
<td>14.2</td>
<td>5.4</td>
<td>18.0</td>
</tr>
<tr>
<td>D</td>
<td>8.9</td>
<td>0.49</td>
<td>11.0</td>
<td>13.9</td>
<td>5.2</td>
<td>17.3</td>
</tr>
<tr>
<td>E</td>
<td>8.9</td>
<td>0.50</td>
<td>11.1</td>
<td>13.6</td>
<td>4.2</td>
<td>14.6</td>
</tr>
<tr>
<td>Mean</td>
<td>8.8</td>
<td>0.49</td>
<td>10.9</td>
<td>13.9</td>
<td>5.0</td>
<td>16.8</td>
</tr>
<tr>
<td>Standard deviation</td>
<td>0.1</td>
<td>0.02</td>
<td>0.4</td>
<td>0.3</td>
<td>0.5</td>
<td>1.3</td>
</tr>
</tbody>
</table>

3.3.2 Column Preparation

The repacked soil columns were prepared by oven drying the soil at 40°C, grinding it, sieving it to obtain the <2 mm fraction and packing it into 4 cm height, 5.4 cm internal diameter Perspex columns. The soils were packed at bulk densities of 1.3 or 1.4 ± 0.002 g/cm³. The desired bulk density was achieved by accurately weighing the prepared soil, transferring it into a set of 2 Perspex columns and then applying pressure to compact the soil into the lower column (Figure 3-3). The average moisture content of the soil used to pack the columns was 3.68% with a standard deviation of 0.04%, based on drying 3 subsamples in an oven at 105°C for 24 hours.
All of the solutions applied to the soil columns were prepared using boiled distilled water to eliminate air. The salts were weighed using an AND™ analytical balance with a precision of ±0.0001 g. The solutions were prepared using either one or two litre volumetric flasks.

A saturated gypsum solution was used to determine the maximum hydraulic conductivity possible in response to gypsum applications. Using a saturated gypsum solution instead of
mixing powdered gypsum into the soil removes the influences of incomplete gypsum dissolution and variations in gypsum placement.

The gypsum solution was made by dissolving $2.100 \pm 0.001\text{g}$ of $\text{CaSO}_4$ in one litre of boiled, deionised water. Literature values for the solubility of $\text{CaSO}_4$ were found to vary substantially. The CRC handbook (Lide 2009) listed the solubility of $\text{CaSO}_4$ in water to be $0.205\text{g} / 100\text{g water}$ which is equivalent to $30.1\text{meq/L}$; the Merck Index (Budavari 1996) listed the solubility to be $0.2\text{parts} / 100\text{parts}$ at $18.75^\circ\text{C}$ which is equivalent to $29.4\text{meq/L}$; and Scotter (1985) conducted experiments using saturated gypsum solutions with $29\text{meq/L}$ of $\text{CaSO}_4$. The gypsum solutions made for these experiments used $2.1\text{g CaSO}_4 / 1\text{L}$ of distilled water, which is equivalent to $30.9\text{ meq/L}$, to ensure that the solution was saturated.

Mixed cation salt solutions were used for comparison with the gypsum solutions and for determination of the influence of irrigation water salinity. All of the mixed cation solutions were prepared using $\text{NaCl}$, $\text{CaCl}_2$ and $\text{MgCl}_2$, in suitable ratios with the quantity of each salt maintained at $\pm 0.003\text{ g}$ of the predetermined mass. The purpose of using these salts was to control the cation composition and salinity of the applied solution. The influence of adding various anions was not considered in these experiments.

A low salinity solution was used as an irrigation water surrogate, for comparison with the saturated gypsum solution. The concentrations of cations in this solution are the same as those measured in a sample of the local Burdekin River water (Table 3-2). The data for the “Burdekin River water” composition was obtained from the Department of Environment and Resource Management. In the Mulgrave area of the BHWSS, water from the Burdekin River is used for irrigation. The sampling site chosen, “Burdekin River at Clare: 120006B”, is close to the Mulgrave area of the BHWSS. The 2008 analysis results were chosen as a representative composition for this site because these results were consistent with the mean and median values for the long term measurements at this site.
Table 3-2. The composition of “Burdekin River water” at site 120006B, concentrations in mg/L.

<table>
<thead>
<tr>
<th></th>
<th>Ca</th>
<th>Mg</th>
<th>Na</th>
<th>K</th>
<th>HCO$_3$</th>
<th>SO$_4$</th>
<th>Cl</th>
<th>EC</th>
</tr>
</thead>
<tbody>
<tr>
<td>2008 analysis</td>
<td>15</td>
<td>7.8</td>
<td>18.0</td>
<td>3.2</td>
<td>101.0</td>
<td>5.5</td>
<td>21.0</td>
<td>232µS/cm</td>
</tr>
<tr>
<td>Mean of analyses from 1975 to 2006</td>
<td>15</td>
<td>7.5</td>
<td>16.8</td>
<td>3.1</td>
<td>96.3</td>
<td>2.9</td>
<td>18.1</td>
<td>218µS/cm</td>
</tr>
</tbody>
</table>

The composition of the mixed cation solutions used to determine the influence of salinity on hydraulic conductivity was based on the soil cation composition. The aim was to maintain the soil exchangeable sodium concentrations so that the influence of salinity could be measured. The exchangeable sodium percentage of the bulk soil collected ranged from 14.6 to 18. The solutions used had a sodium adsorption ratio (SAR) of 13, this is assumed to be equivalent to an ESP of 15 based on the published USDA relationship (USDA 1954)

$$ESP = \frac{100(-0.0126 + 0.01475SAR)}{1 + (-0.0126 + 0.01475SAR)}$$

As the soil contains appreciable quantities of both calcium and magnesium, both of these ions were included in the applied solutions. Potassium was only present in small quantities so this ion was not included in the solutions. The concentrations of calcium and magnesium were kept equal, while maintaining the chosen SAR, because the concentrations of exchangeable calcium and exchangeable magnesium in the soils were fairly similar in the local soils and in the irrigation water used locally. Because there is some variation in exchangeable cation concentrations in the soils, it is expected that this solution may cause small changes in the concentrations of these cations in the soils treated.

The solutions used to determine the impact of salinity on the hydraulic conductivity were a 50 meq/L solution and a 100 meq/L solution. These salt concentrations were chosen because, according to the results of McNeal and Coleman (1966), at low levels of soil sodicity e.g. SAR < 25, the maximum hydraulic conductivity would occur below 100 meq/L. The original experimental plan was to apply the 100 meq/L solution, wait for equilibration then apply the 50 meq/L solution to the same column. However, chemical equilibration did not coincide with stable hydraulic conductivities and extended periods of
leaching were required with these solutions. As a result of this, the different solutions were applied to different columns instead of applying the solutions sequentially.

### 3.3.4 Determination of hydraulic conductivity

Hydraulic conductivity was determined using the constant head method (Klute and Dirksen 1986). The columns were wet from the bottom using the percolating solution (Figure 3-4). The same solution was then applied continuously while the flow rates were measured. The flow rate was determined by measuring the mass and volume of the leachate at regular time intervals. The mass was measured by leaving the solution collection bottle on a calibrated AND™ analytical balance connected to a laptop with specialised software set up to record the mass at ten minute intervals. The volume of leachate was measured at 24 hour intervals using measuring cylinders.

The hydraulic conductivity (K) was calculated using Darcy’s Law using the flow rates, measured at 24 hour intervals. For example, when the measured flow rate was 50 mL/day:

\[
K = \frac{\text{Flow Rate (Q)} \times \text{Length of Column (L)}}{\text{(Cross-sectional Area of Column (A))} \times \Delta h}
\]

\[
= \frac{50 \text{ cm}^3/\text{day} \times 4 \text{ cm}}{22.9 \text{ cm}^2 \times 60 \text{ cm}}
\]

\[= 0.15 \text{ cm/day}\]

(where \(\Delta h\) is the measurement depicted in Figure 3-4).

A constant hydraulic gradient of 20 cm/cm was maintained for all experiments. During preliminary experiments this gradient was determined to be the minimum gradient to enable flow to occur within 48 hours.

The column leaching was continued until stable flow rates were achieved. This required between one and five months of continuous leaching, depending on the applied solution. Multiple criteria were used to define a period of “stable flow rates” based on volume measurements; a) less than three percent variation in the volume between consecutive measurements for at least seven days, b) less than three percent variation in volume between the first and the last measurement over a seven day period of stable flow and c) less than three percent coefficient of variation during the stable period.
It was expected that the flow rates would stabilise when the soil was in equilibrium with the applied solution. Measurements of the leachate salinity and composition were therefore used to assist in determining equilibration with the applied solution.

When analysing the hydraulic conductivity measurements from these experiments, the mean and standard deviation were calculated for each replicate, once stable flow rates were achieved. The mean hydraulic conductivities were used to compare groups of replicates for different treatments. The student’s t-test was applied for these comparisons.
Figure 3-4. Diagram showing column leaching apparatus.
3.3.5 Measurement of leachate chemistry

Samples of the leachate solution were collected at 24 hour intervals. To determine whether equilibration had occurred, the electrical conductivity and cation concentrations in the leachate solutions were measured. The electrical conductivity of each collected leachate solution was measured. The concentration of cations in the leachate solution was determined at approximately 1 week intervals. The cations were analysed using Atomic Absorption Spectrometry (AAS). The pH was also determined for the samples that were used for cation analyses.

The electrical conductivity (EC) was measured using a TPS™ conductivity meter. Calibration was performed using deionised water and a 2.76mS/cm EC standard, in between every three to five sample measurements. The cation and pH analyses were performed at the Natural Resource Sciences laboratories at Indooroopilly, these are NATA accredited laboratories. The QA procedures used for the cation analyses include regular calibration, analysis of blanks and analysis of duplicates.

At the end of the experiments, the wet and dry weights for each soil column were measured. This was to enable the approximate water contents at the end of the experiments to be determined. The concentrations exchangeable cations and the particle size distribution were also determined for each soil column. The purpose of these analyses was to determine the extent of cation exchange that had occurred during leaching and to account for variations in soil particle size distribution between soil columns.
3.4 Results

The purpose of these experiments was to determine the influence of applying gypsum and solutions with varying salinity on the hydraulic conductivity of sodic clay soils from the Mulgrave area of the BHWSS. The results that will be presented in this section include the hydraulic conductivity measurements and the leachate chemistry measurements. The hydraulic conductivity results are presented as time series graphs and statistical summaries of the final stable hydraulic conductivity measurements. The leachate chemistry results are presented as time series. The measured leachate chemistry represents the soil solution chemistry. Changes in the soil solution chemistry were used to determine equilibration with the applied solution and to establish whether cation exchange is occurring.

The results are firstly presented separately for the two sets of experiments which determined the influence of gypsum and salinity respectively. The hydraulic conductivity and leachate chemistry results are plotted against leaching time and pore volumes of leachate. The use of pore volumes for these plots is particularly relevant for the columns that were packed at two different bulk densities.

3.4.1 Results of Preliminary Investigations

The suitability of the experimental equipment used in this study was tested in a number of different ways prior to embarking on the full set of measurements. The first set of column experiments were set up using a saturated gypsum solution and columns packed at 1.3 g/cm$^3$. This experiment was run for three weeks and the flow appeared to stabilise after two weeks. During the last two weeks of the experiment, the columns were observed to be leaking. Tightening of the screws holding the column apparatus together did not stop the leaks. The column apparatus was subsequently modified by adding two sets of rubber rings; this resolved the problem of leaking for further experiments.

The leaking columns were later determined to have had a significant influence on the hydraulic conductivity results. When two more replicates were run using the same conditions with the improved apparatus, the final hydraulic conductivity reached was
significantly greater and the time to a “stable flow rate” was longer, taking around a month (Figure 3-5).

To determine the most suitable measurement interval for the flow rate, data was collected at ten minute intervals using electronic balances and at 24 hour intervals using volume measurements. The calculated hydraulic conductivities based on these measurements were then compared. Although there was a significant amount of scatter with the mass measurements the overall trend and the hydraulic conductivity values were very similar to the daily volume measurements (Figure 3-6).
Figure 3-6. Hydraulic conductivity, calculated using both mass and volume measurements. The dots are based on the mass measurements and the line is based on the volume measurements.

As the changes in flow rate were very slow for these experiments, flow rate measurements at 24 hour intervals were determined to give sufficient information. Mass measurements at ten minute intervals were continued for three columns at a time for the remainder of the experiments, as this was the number of balances available. All remaining columns were run with only 24 hour interval flow rate measurements.

For the measurement with scales, the solution collection bottles were only moved once every 24 hours, to minimize disturbance to the mass measurements. Using this method, the range for the electronic balance was exceeded once the flow rate exceeded 400 mL/day.

Problems were also encountered with the electronic balance measurements when the flow rates were very low, for example, less than 10 mL/day. In these cases, the fluctuations in the measurements were large compared with the actual results.
### 3.4.2 Hydraulic Conductivity Responses to Gypsum Applications

The first experiments using a saturated gypsum solution were conducted using columns packed to 1.4 g/cm$^3$, the maximum bulk density that could be achieved using the method described in section 3.3.2. When a saturated gypsum solution was applied to the soil columns there was gradual increase in hydraulic conductivity. Flow out of the columns took approximately 48 hours to commence after the solution was first applied to the dry soil. A minimum of 3 weeks was taken to reach a stable flow rate for the columns packed at 1.4 g/cm$^3$, with some variation between replicates (Figure 3-7). This corresponded to between 15 and 40 pore volumes of solution passing through the column (Figure 3-8).

![Figure 3-7. Hydraulic conductivity responses to the application of a saturated gypsum solution.](image-url)
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The average stable hydraulic conductivity for the five replicates packed at 1.4 g/cm$^3$ with gypsum applied was 0.147 cm/day with a standard deviation of 0.040 cm/day (Table 3-3).

Table 3-3. Stable hydraulic conductivity measurements in cm/day when a saturated gypsum solution was applied to columns packed at 1.4 g/cm$^3$.

<table>
<thead>
<tr>
<th></th>
<th>A</th>
<th>B</th>
<th>C</th>
<th>D</th>
<th>E</th>
<th>Overall</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mean</td>
<td>0.117</td>
<td>0.124</td>
<td>0.187</td>
<td>0.112</td>
<td>0.194</td>
<td>0.147</td>
</tr>
<tr>
<td>Standard deviation</td>
<td>0.001</td>
<td>0.002</td>
<td>0.004</td>
<td>0.001</td>
<td>0.001</td>
<td>0.040</td>
</tr>
</tbody>
</table>

3.4.3 Hydraulic conductivity: Gypsum vs. low salinity water

The stable hydraulic conductivities reached when the low salinity water (2 meq/L) was applied ranged from 0.004 to 0.007 cm/day (Table 3-4). There was a statistically significant difference, at a probability level of 0.05 between hydraulic conductivities for the gypsum solution and the low salinity water solution. The hydraulic conductivities for the low salinity water were at least ten times less than the hydraulic conductivities when gypsum was applied to soil columns packed at the same bulk density.
Table 3-4. Stable hydraulic conductivity measurements in cm/day when low salinity water or gypsum were applied to columns packed at 1.4 g/cm³.

<table>
<thead>
<tr>
<th></th>
<th>A</th>
<th>B</th>
<th>C</th>
<th>D</th>
<th>E</th>
<th>Mean</th>
<th>Standard deviation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gypsum</td>
<td>0.117</td>
<td>0.124</td>
<td>0.187</td>
<td>0.112</td>
<td>0.194</td>
<td>0.147</td>
<td>0.040</td>
</tr>
<tr>
<td>Low Salinity Water</td>
<td>0.004</td>
<td>0.004</td>
<td>0.005</td>
<td>0.007</td>
<td>0.006</td>
<td>0.005</td>
<td>0.001</td>
</tr>
</tbody>
</table>

The trends in hydraulic conductivity when low salinity water was applied were quite different to the trends when gypsum solutions were applied. When a low salinity mixed cation solution was applied to soil columns the flow rate increased to a maximum before gradually decreasing to a stable flow rate (Figure 3-9 and Figure 3-10). Throughout the measurements, the hydraulic conductivity remained below 0.02 cm/day.

Figure 3-9. Responses in hydraulic conductivity when a low salinity mixed cation solution (2 meq/L) is applied.
3.4.4 Leachate chemistry responses to gypsum applications

The leachate chemistry results in this section are presented using time and volume of solution passed through the columns as independent variables. By using pore volumes as the independent variable, the variability between replicates was reduced resulting in one response function in most cases for the leachate chemistry data.

When the saturated gypsum solution was applied, the electrical conductivities (ECs) of the leachate solutions were initially relatively high with values ranging from 6.70 to 11.34 mS/cm. The ECs then gradually decreased over time (Figure 3-11 and Figure 3-12). After approximately three weeks or 15 pore volumes of solution, the EC of the leachate solution was close to the EC of the gypsum solution applied.

Variations in the cation compositions of the leachate solutions are expected to reflect changes in the exchangeable cation composition of the soil since the composition of the applied solution is constant. For the first cation analyses, 3 days after leaching commenced, the sodium concentration of the leachate was at a maximum (Figure 3-13).
This corresponded with a high initial EC in the leachate solutions (Figure 3-11). At this point, the magnesium concentrations were also at a maximum (Figure 3-17) while the calcium concentrations were > 200 mg/L (Figure 3-15). Between this measurement and the next measurement, 5 to 10 days later, the concentrations of all of the cations decreased. After this point, the sodium concentrations continued to decrease (Figure 3-13) while the calcium concentrations increased (Figure 3-15).

The gradual decrease in sodium corresponding with a gradual increase in calcium indicates that the applied calcium was reducing the sodicity of the soil. By 28 days, after 15 pore volumes of solution had passed through the columns, the sodium concentrations in the leachate solution had been reduced significantly, to less than 10 mg/L. At the end of the experiment, sodium concentrations were close to zero while calcium concentrations were between 440 and 540 mg/L. The concentration of calcium in the applied solution was > 600 mg/L.

Figure 3-11. Electrical conductivity of leachate solutions when a saturated gypsum solution is applied. The red horizontal line shows the average electrical conductivity of the gypsum solution.
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Figure 3-12. Electrical conductivity of leachate solutions when a saturated gypsum solution is applied, solution as a function of pore volumes of solution collected. The red horizontal line shows the average electrical conductivity of the gypsum.

Figure 3-13. Sodium concentrations in the leachate solutions when a saturated gypsum solution was applied.
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Figure 3-14. Sodium concentrations in the leachate solutions when a saturated gypsum solution was applied as a function of pore volumes collected.

Figure 3-15. Calcium concentrations in the leachate solutions when a saturated gypsum solution was applied, where the horizontal dashed line shows the calcium concentration in the gypsum solution.
The magnesium concentrations showed some interesting behaviour. While the initial trend of a high initial concentration followed by a decrease in concentration mimicked the results for the other cations, the concentration of magnesium started to increase again after 10 days before changing trends again and starting to decrease after 23 days (Figure 3-17). The increase in magnesium after day 10 indicates an increase in the rate that exchangeable magnesium was being replaced by the applied calcium. The subsequent decrease in magnesium indicates that the rate of exchange was slowing down. The magnesium concentration at the end of the experiment was still greater than 50 mg/L, so at this point calcium was still replacing the exchangeable magnesium in the soil.

Figure 3-18 shows that the rate of change of magnesium concentrations with respect to pore volumes of leachate collected was similar between the replicates with the exception of column E. For column E, the first magnesium concentration measured was lower than for the other replicates and final magnesium concentration was also lower, though the final measurement was also associated with a greater number of pore volumes.
Reductions in magnesium concentrations in the leachate solutions occurred after the soil sodicity had been significantly reduced. The increases in calcium concentrations that occurred during this period corresponded to magnesium reductions. This trend can be seen clearly using plots of calcium to magnesium ratios (Figure 3-19 and Figure 3-20).

Figure 3-17. Magnesium concentrations in the leachate solutions when a saturated gypsum solution was applied.
Figure 3-18. Magnesium concentrations in the leachate solutions when a saturated gypsum solution was applied as a function of pore volumes collected.

Figure 3-19. Calcium to Magnesium ratio of the leachate solutions when a saturated gypsum solution was applied.
The sodium adsorption ratio (SAR) of the solution is a relationship between the concentration of sodium, calcium and magnesium ions. The SAR of the leachate solutions decreased during the experiment, which represents a decrease in sodium concentrations relative to calcium and magnesium concentrations (Figure 3-21 and Figure 3-22). The decreases in the SAR of the leachate solution during the experiment generally corresponded with increases in measured hydraulic conductivity (Figure 3-23). However, columns C and E show further increases in hydraulic conductivities occurring even when the SAR of the leachate solutions had been reduced to close to zero.
Figure 3-21. Sodium Adsorption Ratio of the leachate solutions when a saturated gypsum solution was applied.

Figure 3-22. Sodium Adsorption Ratio of the leachate solutions when a saturated gypsum solution was applied as a function of pore volumes collected.
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It is expected that equilibration with the applied solution would correspond with stable hydraulic conductivity measurements. Figure 3-24 shows that stable hydraulic conductivity measurements corresponded with stable electrical conductivity measurements. However, the decreases in EC after 10 days of leaching were relatively small (20%) compared with the significant increases in hydraulic conductivity (60%). Therefore determining when equilibration has occurred using EC measurements alone may be difficult.

Figure 3-23. Sodium Adsorption Ratio of the leachate solutions versus hydraulic conductivity when a saturated gypsum solution was applied.
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Figure 3-24. EC of the leachate solution vs. hydraulic conductivity for column B.

The decreases in the EC of the leachate solutions correlate with decreases in the SAR of the leachate solutions (Figure 3-25). The final EC was around 2.1 mS/cm and this corresponded to SARs of less than 5. However, it is important to note that further increases in hydraulic conductivity occurred as the SAR was reduced below 5, towards zero (Figure 3-23) and the decreases in EC at this point were difficult to detect.

Figure 3-25. Sodium Adsorption Ratio versus EC of the leachate solutions when a saturated gypsum solution was applied.
3.4.5 The Influence of Bulk Density on Hydraulic Conductivity when Gypsum is Applied

When the columns were packed to 1.3 g/cm³ and a saturated gypsum solution was applied, there was gradual increase in hydraulic conductivity (Figure 3-26 and Figure 3-27), as there was for the columns packed at 1.4 g/cm³. The time taken to reach the maximum hydraulic conductivity was similar for the two bulk densities but there were differences in the maximum hydraulic conductivity, the variability between replicates and the volume of solution applied.

Figure 3-26. Hydraulic conductivity responses to the application of a saturated gypsum solution to soil columns packed at a bulk density of 1.3 g/cm³.
Figure 3-27. Hydraulic conductivity responses to the application of a saturated gypsum solution to soil columns packed at a bulk density of 1.3 g/cm$^3$, as a function of pore volumes collected.

The average stable hydraulic conductivity for the five replicates packed at 1.3g/cm$^3$ was 0.964 cm/day with a standard deviation of 0.537 cm/day (Table 3-5). The hydraulic conductivity for the columns packed at 1.3 g/cm$^3$ is significantly higher, at a probability level of 0.05, than the hydraulic conductivity for the columns packed at 1.4 g/cm$^3$. The standard deviation of the hydraulic conductivities for the columns packed at the lower bulk density was more than ten times greater than the corresponding standard deviation for the higher bulk density (Table 3-5 and Figure 3-28). For the 1.3 g/cm$^3$ columns, 150 to 250 pore volumes of solution were applied before the hydraulic conductivity stabilised (Figure 3-27), compared with 15 to 40 pore volumes for the 1.4 g/cm$^3$ columns (Figure 3-8).

Table 3-5. Stable hydraulic conductivity measurements in cm/day when gypsum was applied to columns packed at 1.3 and 1.4 g/cm$^3$.

<table>
<thead>
<tr>
<th>Gypsum, 1.4g/cm$^3$</th>
<th>A</th>
<th>B</th>
<th>C</th>
<th>D</th>
<th>E</th>
<th>mean</th>
<th>standard deviation</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0.117</td>
<td>0.124</td>
<td>0.187</td>
<td>0.112</td>
<td>0.194</td>
<td>0.147</td>
<td>0.040</td>
</tr>
<tr>
<td>Gypsum, 1.3g/cm$^3$</td>
<td>0.889</td>
<td>1.450</td>
<td>0.244</td>
<td>1.534</td>
<td>0.702</td>
<td>0.964</td>
<td>0.537</td>
</tr>
</tbody>
</table>
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Figure 3-28. The range in hydraulic conductivity vs. bulk density of the packed soil columns.

The hydraulic conductivity for intact cores with gypsum applied was only able to be measured for one column. There was no flow from the second intact core. The measured hydraulic conductivity for the intact core was 0.001 cm/day, with the flow rate remaining constant for a two week period.

3.4.6 The Influence of Bulk Density on Leachate Chemistry when Gypsum is Applied

The leachate chemistry for the columns packed at 1.3 g/cm$^3$ showed similar trends to the columns packed at 1.4 g/cm$^3$ when the results were plotted against time. For example, the time taken for EC and sodium concentration to decrease to a minimum for the columns packed at 1.3 g/cm$^3$ was between 20 and 25 days (Figure 3-29 and Figure 3-30) this time period was approximately the same as for the columns packed at 1.4 g/cm$^3$. However, there were differences in the initial and final concentrations of the cations. The concentrations of the cations measured after three days of leaching were much lower for the columns packed at the lower bulk density (Figure 3-30, Figure 3-31 and Figure 3-32).
Figure 3-29. Electrical conductivity of the leachate solutions when gypsum was applied to columns packed at 1.3 g/cm$^3$.

Figure 3-30. Sodium concentrations in the leachate solutions when a saturated gypsum solution was applied to columns packed at 1.3 g/cm$^3$. 
Figure 3-31. Calcium concentrations in the leachate solutions when a saturated gypsum solution was applied to columns packed at 1.3g/cm$^3$.

Figure 3-32. Magnesium concentrations in the leachate solutions when a saturated gypsum solution was applied to columns packed at 1.3g/cm$^3$. 
The similarities between the leachate chemistry for the columns packed at different bulk densities were more apparent when the changes in leachate chemistry against pore volumes of solution collected were plotted for both bulk densities. When the soil was packed at 1.3 g/cm$^3$ instead of 1.4 g/cm$^3$, the time taken to reach a stable hydraulic conductivity was similar but the number of pore volumes was approximately four times larger. For both bulk densities, sodium was reduced to a minimum after 15 pore volumes (Figure 3-33 and Figure 3-34). After this point, there was essentially no change in sodium concentrations. However, there were differences in the calcium and magnesium concentrations after this point. The final calcium concentrations were higher for the columns packed at 1.3 g/cm$^3$ than for the columns packed at 1.4 g/cm$^3$ (Figure 3-35). Conversely, the final magnesium concentrations were lower for the columns packed at 1.3 g/cm$^3$ than for the columns packed at 1.4 g/cm$^3$ (Figure 3-36). These differences can be seen clearly when the calcium to magnesium ratio is plotted (Figure 3-37).

![Figure 3-33. Sodium concentration in the leachate solutions for two bulk densities.](image)
Figure 3-34. Sodium Adsorption Ratio of the leachate solutions for two bulk densities.

Figure 3-35. Calcium concentration in the leachate solutions for two bulk densities.
Figure 3-36. Magnesium concentration in the leachate solutions for two bulk densities.

Figure 3-37. Calcium to Magnesium ratio of the leachate solutions for two bulk densities.
3.4.7 Final soil properties for columns that had gypsum applied

The moisture content, exchangeable cation concentrations and particle size distribution were measured for each of the soil columns that had gypsum applied. The particle size distribution was measured to determine whether the clay content had a significant influence on the final hydraulic conductivity. There was very little variation in clay percentage between the columns for each bulk density, but there were significant negative correlations between clay percentage and hydraulic conductivity for each bulk density. The $R^2$ values for these correlations were 0.73 and 0.38 for the columns packed to 1.3 g/cm$^3$ and 1.4 g/cm$^3$ respectively.

Leaching the soils with a saturated gypsum solution lead to a reduction in exchangeable sodium from approximately 5 meq/100g to < 0.6 meq/100g (Table 3-6), equivalent to an ESP of < 1.5. The exchangeable magnesium concentrations decreased from approximately 13 to < 7 meq/100g (Table 3-6). The lowest final exchangeable magnesium concentrations for each bulk density corresponded with the highest final hydraulic conductivity and the highest final exchangeable magnesium concentrations for each bulk density corresponded with the lowest final hydraulic conductivity as shown in Table 3-6 and Figure 3-38. The negative correlations between the final exchangeable magnesium concentrations and the final hydraulic conductivity resulted in $R^2$ values of 0.69 and 0.85 for the columns packed to 1.3 g/cm$^3$ and 1.4 g/cm$^3$ respectively.
Table 3-6. Final properties for soil columns treated with gypsum, where the highest hydraulic conductivity results are shown in bold and the lowest hydraulic conductivity results are underlined.

<table>
<thead>
<tr>
<th>Bulk density / replicate</th>
<th>Final Moisture Content (g/g)</th>
<th>Final exchangeable cation concentrations (meq/100g)</th>
<th>Sand %</th>
<th>Silt %</th>
<th>Clay %</th>
<th>Final K (cm/day)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Sodium</td>
<td>Calcium</td>
<td>Magnesium</td>
<td></td>
<td></td>
</tr>
<tr>
<td>1.3 / A</td>
<td>0.32</td>
<td>0.55</td>
<td>32.4</td>
<td>0.45</td>
<td>18</td>
<td>20</td>
</tr>
<tr>
<td>1.3 / B</td>
<td>0.33</td>
<td>0.50</td>
<td>35.7</td>
<td>0.38</td>
<td>19</td>
<td>20</td>
</tr>
<tr>
<td>1.3 / C</td>
<td>0.30</td>
<td>0.50</td>
<td>35.7</td>
<td>&lt;0.31</td>
<td>20</td>
<td>19</td>
</tr>
<tr>
<td>1.3 / D</td>
<td>0.32</td>
<td>0.53</td>
<td>36.7</td>
<td>0.43</td>
<td>20</td>
<td>17</td>
</tr>
<tr>
<td>1.4 / A</td>
<td>0.30</td>
<td>0.38</td>
<td>26.3</td>
<td>5.36</td>
<td>16</td>
<td>17</td>
</tr>
<tr>
<td>1.4 / B</td>
<td>0.30</td>
<td>0.37</td>
<td>25.2</td>
<td>4.73</td>
<td>18</td>
<td>13</td>
</tr>
<tr>
<td>1.4 / C</td>
<td>0.30</td>
<td>0.37</td>
<td>28.2</td>
<td>3.27</td>
<td>14</td>
<td>14</td>
</tr>
<tr>
<td>1.4 / D</td>
<td>0.29</td>
<td>0.52</td>
<td>33.8</td>
<td>1.31</td>
<td>19</td>
<td>18</td>
</tr>
</tbody>
</table>

Figure 3-38. Final exchangeable magnesium concentrations versus final hydraulic conductivity.
3.4.8 Hydraulic conductivity responses to application of mixed cation salt solutions

The trends in hydraulic conductivity when 50 meq/L and 100 meq/L mixed cation salt solutions were applied showed an increase to a maximum value during the first 2 to 4 weeks, followed by a gradual decrease, during the subsequent 3 to 6 months, or up to 800 pore volumes, before a stable hydraulic conductivity was reached (Figure 3-39 and Figure 3-40). The maximum hydraulic conductivities for the 100 meq salt solution ranged from 1.18 cm/day to 2.20 cm/day whereas for the 50 meq salt solution the range was 0.62 cm/day to 0.71 cm/day. The final hydraulic conductivities for the 100 meq solution ranged from 0.24 to 0.40 cm/day whereas the final hydraulic conductivities for the 50 meq solution ranged from 0.13 to 0.20 cm/day. The increases in maximum hydraulic conductivity corresponded with increases in the leaching time needed to reach a stable hydraulic conductivity (Figure 3-39).

There were a few obvious differences between the 50 meq solution results and the 100 meq solutions results besides the differences in the maximum and final hydraulic conductivities. There was more variation between replicates for the 100 meq solution and the time taken to reach the maximum hydraulic conductivity and was longer for the 50 meq solution than for the 100 meq solution (Figure 3-39) but the number of pore volumes was almost the same (Figure 3-40). The time taken to reach the final stable hydraulic conductivity was however longer for the 100 meq solution than for the 50 meq solution.
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![Graph showing hydraulic conductivity responses](image1)

**Figure 3-39.** Hydraulic conductivity responses to the application of mixed cation solutions to soil columns packed at a bulk density of 1.3 g/cm³.

![Graph showing hydraulic conductivity responses](image2)

**Figure 3-40.** Hydraulic conductivity responses to the application of mixed cation solutions to soil columns packed at a bulk density of 1.3 g/cm³ as a function of pore volumes collected.

### 3.4.9 Leachate chemistry responses to application of mixed cation salt solutions

The EC of the leachate solutions collected when mixed cation salt solutions were applied to soil columns was fairly stable, after the initial decrease in the EC (Figure 3-41 and Figure 3-42, where the EC of the applied solutions is shown as solid lines). However the results do show some small fluctuations over time, particularly for the 100 meq solution.
Figure 3-41. Electrical conductivity of leachate solutions when mixed cation solutions were applied.

Figure 3-42. Electrical conductivity of leachate solutions when mixed cation solutions were applied, as a function of pore volumes collected.

The cation concentrations of the leachate solutions collected when mixed cation solutions were applied were fairly constant, after the initial increase or decrease (Figure 3-43, Figure 3-44, Figure 3-45, Figure 3-46, Figure 3-47 and Figure 3-48). This initial change in concentration is likely to reflect differences between the soil exchangeable cation concentration and the relative concentrations of cations in the applied solutions. After this initial change, the cation concentrations were generally close to those of the applied solutions (Figure 3-43, Figure 3-44, Figure 3-45, Figure 3-46, Figure 3-47 and Figure 3-48).
Out of the cation concentrations, the sodium concentrations were the least stable. Small fluctuations in sodium concentrations occurred throughout the leaching period with variations in the trends between replicates. Several of the replicates exhibited reductions in sodium concentration towards the end of the leaching period but these reductions were generally within the range of the concentrations exhibited throughout the leaching period.

The pH of the leachate solutions also showed an initial decrease with fluctuations in pH after this point < ±0.5 pH units (Figure 3-49 and Figure 3-50). The pH of the applied solutions was slightly acidic (< 6) but the pH of the leachate solutions was generally greater than 6.2 and often closer to 7.

Unlike the changes in hydraulic conductivity when mixed cation solutions were applied which occurred throughout the leaching period, most of the changes in cation concentrations occurred during the early stages of leaching. As a result, the differences in the number of pore volumes that passed through the columns for different replicates did not correspond with differences in leachate chemistry. Instead, the leachate chemistry closely reflects the composition of the applied solutions. This differs from the results for gypsum solutions applied to columns packed at different bulk densities, where the changes in leachate chemistry were closely associated with the number of pore volumes that passed through the columns.
Figure 3-43. Calcium concentrations in the leachate solutions when mixed cation solutions were applied.

Figure 3-44. Calcium concentrations in the leachate solutions when mixed cation solutions were applied, as a function of pore volumes collected.
Figure 3-45. Magnesium concentrations in the leachate solutions when mixed cation solutions were applied.

Figure 3-46. Magnesium concentrations in the leachate solutions when mixed cation solutions were applied, as a function of pore volumes collected.
Figure 3-47. Sodium concentrations in the leachate solutions when mixed cation solutions were applied.

Figure 3-48. Sodium concentrations in the leachate solutions when mixed cation solutions were applied, as a function of pore volumes collected.
Figure 3-49. pH of the leachate solutions when mixed cation solutions were applied.

Figure 3-50. pH of the leachate solutions when mixed cation solutions were applied, as a function of pore volumes collected.
3.4.10 Final soil properties for columns that had mixed cation solutions applied

To determine whether the differences in hydraulic conductivity and leachate chemistry between replicates could be accounted for by differences in soil texture or final cation concentrations, these properties were measured for each of the columns (Table 3-7). The maximum hydraulic conductivity reached was for column C with the 100 meq/L solution. The clay content was slightly lower for this replicate but only by 1% and the final sodium and magnesium concentrations were similar to the other columns. In fact, for the columns with the 100 meq/L solution applied, the lowest final sodium concentration corresponded to the lowest hydraulic conductivity. This result was unexpected as the experiments with gypsum solutions applied showed increasing hydraulic conductivity with decreasing sodicity. The final moisture content for all of the replicates only varied slightly but the higher moisture contents corresponded with higher final hydraulic conductivities.

Table 3-7. Final soil properties for columns leached with mixed cation solutions.

<table>
<thead>
<tr>
<th>Salt concentration of applied solution / Replicate</th>
<th>Final Moisture Content (g/g)</th>
<th>Final Exchangeable Cation Concentrations (meq/100g)</th>
<th>Sand %</th>
<th>Silt %</th>
<th>Clay %</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Sodium                Calcium            Magnesium</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>50 meq/L / A</td>
<td>0.32</td>
<td>4.35                  16.8              11.7</td>
<td>17</td>
<td>17</td>
<td>68</td>
</tr>
<tr>
<td>50 meq/L / B</td>
<td>0.32</td>
<td>4.45                  16.2              12.9</td>
<td>17</td>
<td>19</td>
<td>68</td>
</tr>
<tr>
<td>50 meq/L / C</td>
<td>0.32</td>
<td>4.33                  16.5              12.6</td>
<td>18</td>
<td>19</td>
<td>68</td>
</tr>
<tr>
<td>100 meq/L / A</td>
<td>0.32</td>
<td>3.08                  15.2              11.0</td>
<td>16</td>
<td>20</td>
<td>67</td>
</tr>
<tr>
<td>100 meq/L / B</td>
<td>0.33</td>
<td>4.54                  17.8              11.8</td>
<td>17</td>
<td>20</td>
<td>68</td>
</tr>
<tr>
<td>100 meq/L / C</td>
<td>0.33</td>
<td>4.44                  15.7              11.9</td>
<td>20</td>
<td>19</td>
<td>66</td>
</tr>
</tbody>
</table>

As the leachate chemistry trends did not correspond with the observed trends in hydraulic conductivity, i.e. a sharp increase followed by a very gradual decline, the clay mineralogy of two of the columns from this experiment was measured to determine whether there were observable changes in clay mineralogy. The results for these two columns were compared with the results for the bulk soil (Table 3-8). The soils from the columns were found to have slightly lower proportions of illite, higher proportions of kaolinite and lower proportions of smectite, however, these changes were not significant.
Table 3-8. Clay mineralogy for columns leached with mixed cation solutions.

<table>
<thead>
<tr>
<th>Clay mineral</th>
<th>Bulk soil - average</th>
<th>Bulk soil – standard deviation</th>
<th>100 meq/L - C</th>
<th>100 meq/L - D</th>
</tr>
</thead>
<tbody>
<tr>
<td>illite</td>
<td>19.3</td>
<td>3.2</td>
<td>17.8</td>
<td>15.5</td>
</tr>
<tr>
<td>kaolinite</td>
<td>69.6</td>
<td>4.0</td>
<td>74.5</td>
<td>78.3</td>
</tr>
<tr>
<td>smectite</td>
<td>9.9</td>
<td>1.5</td>
<td>7.7</td>
<td>6.2</td>
</tr>
</tbody>
</table>

3.4.11 The Influence of Salinity and Sodicity on Hydraulic Conductivity

A model depicting the influence of salinity and sodicity on hydraulic conductivity on a range of soils was presented by McNeal (1968). These equations, that relate soil swelling and decreases in hydraulic conductivity to soil sodicity and salinity, were described in section 3.2.1.

The general equation needed to predict the changes in hydraulic conductivity with changes in solution chemistry is as follows (McNeal 1968):

\[ 1 - y = cx^n / (1 + cx^n) \]  

(3.1)

where \( y \) is the relative soil hydraulic conductivity, \( c \) and \( n \) are constants for a given soil within a specified range of ESP values and \( x \) is a swelling factor (the calculated interlayer swelling of soil montmorillonite) given by:

\[ x = (f_{mont})(3.6 \times 10^{-4})(ESP^*)(d^*) \]  

(3.2)

where \( f_{mont} \) is the weight fraction of montmorillonite in the soil, ESP* is the adjusted ESP (ESP* = soil ESP – (1.24 + 11.63 log \( C_0 \))), \( d^* \) is the adjusted interlayer spacing (\( d^* = 0 \) (for \( C_0 > 300 \) meq/L), \( d^* = 356.4 \left(C_0^{1/2} + 1.2 \right) \) (for \( C_0 < 200 \) meq/L)), \( C_0 \) is the total salt concentration of the ambient solution.

As a first approximation, acceptable \( n \) values are \( n = 1 \) for ESP < 25, \( n = 2 \) for ESP >25 & <50 and \( n = 3 \) for ESP >50. Only values of \( c \) need to vary between different soils (McNeal 1968). The soil specific constant “c” can be calculated once the hydraulic conductivity has been measured at two salt concentrations, at the same SAR.

As the hydraulic conductivity varied over time when mixed cation solutions were applied, the maximum hydraulic conductivity values are used for illustrative purposes.
Table 3-9. Calculations based on maximum hydraulic conductivity with mixed cation solutions applied.

<table>
<thead>
<tr>
<th>Calculation</th>
<th>Result</th>
</tr>
</thead>
<tbody>
<tr>
<td>Maximum hydraulic conductivity for 100 meq/L solutions</td>
<td>1.77 cm/day</td>
</tr>
<tr>
<td>= (1.18 cm/day + 1.93 cm/day + 2.20 cm/day)/3</td>
<td></td>
</tr>
<tr>
<td>Maximum hydraulic conductivity for 50 meq/L solutions</td>
<td>0.66 cm/day</td>
</tr>
<tr>
<td>= (0.70 cm/day + 0.67 cm/day + 0.62 cm/day)/3</td>
<td></td>
</tr>
<tr>
<td>y (relative hydraulic conductivity)</td>
<td>0.37</td>
</tr>
<tr>
<td>= 0.66 cm/day / 1.77 cm/day</td>
<td></td>
</tr>
<tr>
<td>d* (adjusted interlayer spacing)</td>
<td>4.764</td>
</tr>
<tr>
<td>= 356.4 x C_0^{1/2} (meq/L) + 1.2</td>
<td></td>
</tr>
<tr>
<td>= 356.4 x (50)^{1/2} + 1.2</td>
<td></td>
</tr>
<tr>
<td>ESP* (adjusted ESP)</td>
<td>-4.199</td>
</tr>
<tr>
<td>= ESP - 1.24 + 11.63 log C_0 (meq/L)</td>
<td></td>
</tr>
<tr>
<td>= 16.8 - 1.24 + 11.63 log 50</td>
<td></td>
</tr>
<tr>
<td>x (interlayer swelling)</td>
<td>-0.00072</td>
</tr>
<tr>
<td>= 0.1 x 0.00036 x ESP^* x d^*</td>
<td></td>
</tr>
<tr>
<td>= 0.1 x 0.00036 x -4.199 x 4.764</td>
<td></td>
</tr>
</tbody>
</table>

Using the calculated values for the relative hydraulic conductivity (y) and interlayer swelling (x) (Table 3-9), c can be calculated for different values of “n” using equation 3.1. The results of these calculations are as follows.

When “n” = 1, c = 1.703/-0.00072 = -2364
When “n” = 2, c = 1.703/(-0.00072)^2 = 3283176
When “n” = 3, c = 1.703/(-0.00072)^3 = -4559023996

The c values calculated for the data from the experiments were several orders of magnitude larger than the c values presented in the McNeal (1968) paper. In the cases where n was equal to 1 or 3, the calculated c values were negative whereas the published values are all positive. These values of c and n were used to plot the expected variations in hydraulic conductivity with variations in salt concentration and sodicity (Figure 3-51). The trends in this graph do not show the expected trends, with the exception of the ESP = 5 curve, as trends of decreasing hydraulic conductivity with increasing salt concentration are occurring. This contrasts with the published curves (Figure 3-52). This figure also shows the measured hydraulic conductivity values from this Chapter. The extrapolated curve based on the position of these points is outside of the range of the published curves for ESPs ranging from 15 to 100 even though the ESP of the soil used for the experiments in this Chapter had an ESP of approximately 16.9 (Figure 3-52).
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Figure 3-51. The effect of sodicity and salt concentration on hydraulic conductivity, based on the McNeal (1968) equation.

Figure 3-52. Comparison of published results (McNeal 1968) with the results from this Chapter.
3.4.12 Quality Assurance / Error Analysis

The hydraulic conductivity results presented in this Chapter were based on measurements of the volume of leachate solution collected at 24 hour intervals. The primary sources of error in these measurements include evaporation of the sample during the collection period, incomplete transfer of the collected solutions to the measuring cylinders, measurement errors associated with determining volume using the measuring cylinders and measurement errors associated with determining the gradient. The likely magnitude of these errors was estimated (Table 3-10).

<table>
<thead>
<tr>
<th>Source of Error</th>
<th>Magnitude of Error &amp; Method of Determination</th>
</tr>
</thead>
<tbody>
<tr>
<td>Evaporation of sample</td>
<td>Measured evaporation rate = 1 mL over 10 days</td>
</tr>
<tr>
<td></td>
<td>(0.1 mL approx may evaporate during a 24 hr period)</td>
</tr>
<tr>
<td>Incomplete transfer of sample</td>
<td>1 mL approx may be left in sample collection bottle</td>
</tr>
<tr>
<td>Measurement errors – measuring cylinders</td>
<td>+/- 0.5 mL / 100 mL</td>
</tr>
<tr>
<td></td>
<td>(max error = +/- 3 mL)</td>
</tr>
<tr>
<td>Measurement errors – gradient</td>
<td>+/- 2 cm</td>
</tr>
</tbody>
</table>

The experiments were conducted in controlled laboratory conditions with the temperature in the laboratory recorded at hourly intervals during the experiments. Temperature generally varied by less than 1°C in any one month but over a 5 month period, there was a decline in the temperature measured from between 23 and 23.5°C to between 22 and 22.5°C (Figure 3-53).
3.5 Discussion

The hydraulic conductivity of BHWSS sodic clays was found to increase significantly when a saturated gypsum solution was applied, compared with when a low salinity solution was applied. The measured hydraulic conductivity increased gradually during the period that the soil column was leached with a saturated gypsum solution. The maximum hydraulic conductivity took at least three weeks of continuous leaching to achieve.

Previous published experiments (Mace and Amrhein 2001; Prather, Goertzen et al. 1978), have not continued treatment until the sodicity was reduced to a minimum, thus potentially underestimating the maximum effects possible due to continued gypsum applications. Whereas the initial changes in hydraulic conductivity are likely to be mostly due to electrolyte effects, continued applications of gypsum would also reduce the soil sodicity which is expected to further increase hydraulic conductivity.
The leachate solution chemistry was measured to determine whether equilibration with the applied solution had occurred. The composition of the leachate solutions was expected to be influenced by the initial soil chemistry as well as the composition of the applied solution. At the beginning of the leaching period, the composition of the leachate solutions was more strongly influenced by the initial soil chemistry but as the leaching continued, the chemistry of the leachate solutions increasingly reflected the chemistry of the applied solutions.

When the saturated gypsum solutions were applied, the initial EC of the leachate solutions was > 6 mS/cm, significantly higher than the EC of the applied solutions. The high salt content of these initial leachate solutions probably originated from the combination of the CaSO$_4$ in the applied solution and the salts present in soil used in columns. Salt release due to mineral dissolution, particularly for soils containing calcite (CaCO$_3$), has been found to reduce the susceptibility to hydraulic conductivity reductions when low salinity waters are applied (Shainberg, Rhoades et al. 1981b). This process may also have contributed to the high initial salt content in these experiments.

The gradual decrease in EC before the EC of the applied solution is reached suggests that the leaching of salts contained in the soil, and possibly salt release due to mineral dissolution, occurred at the beginning of the leaching period. It may be possible to use the EC of the leachate solutions to determine whether equilibration with the applied solution has occurred. However, a problem with this approach is that the decreases in EC for the majority of the experiment, after the first few days of leaching, were very gradual relative to the errors associated with measuring EC.

The gradual change in cation composition during the leaching period would also influence the leachate EC. The time taken for the EC to stabilise was approximately equivalent to the time taken for the sodium concentrations in the leachate solutions to stabilise. Changes in calcium and magnesium concentrations in the leachate solutions after sodium concentrations had stabilised did not lead to further changes in EC. Thus measurement of cation concentrations in the leachate solutions was still needed to confirm whether equilibration with the applied solution had occurred.
The trends in leachate cation concentrations indicate that the applied calcium ions replaced the soil exchangeable sodium and magnesium during the leaching period. As the SAR of the leachate solutions, and hence the soil sodicity, was reduced, the hydraulic conductivity increased to a maximum. Even when the SAR of the leachate solutions was below five, further decreases in Na corresponded with further increases in hydraulic conductivity. Although USDA (1954) guidelines for sodic soils suggest an ESP of 15 as the critical value above which soil structure could be deleteriously affected, Northcote and Skene (1972) and McIntyre (1979) have recommended lower critical values for Australian soils of 6 and 5 respectively. The results from these experiments indicate that even lower levels of sodicity can cause soil hydraulic conductivity to be reduced for these Lower Burdekin sodic clay soils.

At the end of the experiments where saturated gypsum solutions were applied, the ESP of the soils was reduced to below 1.5. It is not known whether further leaching would have reduced the ESP even further, but if further decreases in ESP did occur these decreases would not be expected to result in further increases in hydraulic conductivity. For the columns that reached a maximum hydraulic conductivity quickly, for example column D which was packed to 1.4g/cm$^3$, further reductions in the SAR of the leachate solutions occurred, from 1.31 to 0.17, but the hydraulic conductivity did not increase any further.

The calcium ions contained in the applied gypsum solutions reduced the sodicity of the soil while also reducing the exchangeable magnesium concentrations in the soil. There is some debate in the literature about the influence of exchangeable magnesium on hydraulic conductivity (Aydemir and Najjar 2005). McNeal (1968) and Levy (1988) both found that soils high in exchangeable magnesium had a lower hydraulic conductivities than soils high in exchangeable calcium. Levy (1988) and Alperovitch (1981) both suggested that the specific effect of exchangeable magnesium in decreasing hydraulic conductivity results from clay dispersion. In some soils, for example calcareous soils and kaolinitic soils, exchangeable magnesium has no specific effect on hydraulic conductivity (Levy, Shainberg et al. 1997).

A couple of observations about the Ca-Mg exchange that occurred when gypsum was applied in this study warrant further discussion. Firstly, the rate of Ca-Mg exchange
appeared to vary during the leaching period, resulting in fluctuations in the magnesium concentrations in the leachate solution. Secondly, there were significant concentrations of magnesium in the leachate solutions at the end of the experiment. And thirdly, the highest hydraulic conductivity corresponded with the lowest final exchangeable magnesium concentrations and the highest final exchangeable calcium concentrations.

It appears that during the first two weeks of leaching, the applied calcium was predominantly replacing exchangeable sodium. After this point, further applied calcium was replacing the exchangeable magnesium and this resulted in an increase in the magnesium concentrations in the leachate solutions.

The presence of significant concentrations of magnesium in the leachate solutions at the end of the experiment indicates that the applied calcium was still replacing the exchangeable magnesium. It is difficult to separate the effects of exchangeable sodium decreases from the effects of exchangeable magnesium decreases, as both were occurring as the hydraulic conductivity increased.

The fact that the highest measured hydraulic conductivity corresponded with the lowest final exchangeable magnesium concentrations suggests that further reductions in exchangeable magnesium may further increase hydraulic conductivity. However, column D which was packed to 1.4 g/cm$^3$, reached a maximum hydraulic conductivity quickly and further reductions in the magnesium concentrations in the leachate solutions from day 20 onwards did not result in further increases in hydraulic conductivity.

For the experiments where gypsum was applied, five replicates were used for each bulk density and there was substantial variation between the hydraulic conductivity results for the five replicates. The variations relate to the maximum hydraulic conductivity and the time taken to reach the maximum hydraulic conductivity. There was also greater variation between replicates when the columns were packed at 1.3 g/cm$^3$ compared with when the columns were packed at 1.4 g/cm$^3$.

The causes for variations in hydraulic conductivity between replicates are difficult to determine. Factors that may contribute to the variations in final stable hydraulic
conductivity between the replicates include minor differences in particle size distribution and exact bulk density. There were significant negative correlations between clay percentage and hydraulic conductivity for each bulk density. The $R^2$ values for these correlations were 0.73 and 0.38 for the columns packed to 1.3 g/cm$^3$ and 1.4 g/cm$^3$ respectively. The repacked columns were packed to within 0.001 g/cm$^3$ of the desired bulk density for the gypsum experiments. Perhaps as a result of this minor variation in the exact bulk density, there was no consistent correlation between the exact bulk density and the final hydraulic conductivity.

Other reasons for the variability between the results for replicate columns may include differences in compaction patterns or small differences in the initial soil chemistry, particularly the exchangeable cation concentrations and the cation exchange capacities. As the initial soil column conditions including compaction patterns and variations in soil chemistry are not measured, it is not possible to determine whether these factors are responsible.

The reason for greater variations in the final hydraulic conductivity for the 1.3g/cm$^3$ columns compared with the 1.4g/cm$^3$ columns may be the differences in pore size distribution. There is likely to be a greater proportion of larger pore sizes in the columns packed at a lower bulk density and there may also be more variability in the pore sizes.

The bulk density of the packed columns was found to have a significant effect on hydraulic conductivity for the columns where a saturated gypsum solution was applied. Increasing the bulk density of the packed columns corresponded to reductions in the final hydraulic conductivity. This could easily be attributed to the decreased porosity when the soil is more densely packed.

Decreases in bulk density might be expected to increase the extent of cation exchange by increasing the volume of pore space and hence the volume of solution that passes through the columns within a given period of time. In a study related to the measurement of cation exchange capacity in structured soils (Hartmann, Grasle et al. 1998), the accessibility of the exchange surfaces was found to be mainly influenced by the percolation time and the hydraulic conductivity. In this study, there were differences in the cation trends of the
leachate solutions between the two bulk densities. While sodium concentrations showed very similar results for both bulk densities with concentrations reduced to a minimum after approximately 25 pore volumes, the magnesium concentrations were decreased further with additional pore volumes of solution for the lower bulk density. For the columns packed at the lower bulk density, decreases in magnesium concentrations relative to calcium concentrations, were occurring after the point where 25 pore volumes of solution had passed through the columns. Up until this point, the leachate chemistry and hydraulic conductivities showed similar trends for both bulk densities. After 25 pore volumes of solution had passed through the columns, the hydraulic conductivities for the columns with the lower bulk densities increased significantly at the same time as the magnesium concentrations in the leachate solution were increasing relative to calcium concentrations.

The final exchangeable cation concentrations for the soil columns support the theory that magnesium reductions were contributing to higher hydraulic conductivities. This finding follows the results of McNeal, Layfield et al. (1968) and Levy, Vanderwatt et al. (1988) who found that exchangeable magnesium can lead to reductions in hydraulic conductivities in certain soils. However, McNeal, Layfield et al. (1968) stated that the differences between the effects of magnesium and calcium were negligible when comparison were made at equivalent ESPs and that calcium and magnesium can be considered as interchangeable cations when dealing with soil hydraulic conductivity changes under high sodium, low salt conditions. The results in this study showed that for both bulk densities, lower final magnesium concentrations corresponded to higher final hydraulic conductivities. This indicates that hydraulic properties of sodic clay soils in the Lower Burdekin may be sensitive to exchangeable magnesium concentrations, specifically that the presence of exchangeable magnesium may cause reductions in hydraulic conductivity.

As the bulk density of the intact cores collected from the field was significantly higher than the bulk density of the packed columns, much lower hydraulic conductivities would be expected in the field compared with those measured in the laboratory. However, it is possible that other factors may increase hydraulic conductivities in the field, particularly preferential flow through cracks, root holes and higher permeability zones (Crescimanno and De Santis 2004; Scanlon, Tyler et al. 1997). In addition to the impacts of macroporosity, the multi-component nature of soils which can include materials bonded
together into aggregates by various cementing agents and covered with organic and inorganic coatings, may dictate hydraulic conductivity in field soils (Sumner 1993). Intact cores and/or field experiments would be needed in order to incorporate the impacts of these factors.

It is predicted that the trends observed in the laboratory when gypsum solutions were applied, sodicity reductions and increases in hydraulic conductivity, would also occur in the field. The big difference lies in the soil properties and irrigation conditions. In the field, with lower hydraulic conductivities, longer time would be needed for the treatments to flow through the soil profile. In the field, gypsum treatments would take place in combination with irrigation water of varying quality and seasonal rainfall. The proportion of calcium ions in the soil solution would therefore be significantly less than for the conditions in the laboratory where saturated gypsum solutions were applied. As a result, large reductions in the sodicity of the bulk soil in the field would not be expected to occur without continued gypsum additions. Instead, the applied gypsum in the field would be expected to result in only partially reclamation of the sodic soils and the increases in hydraulic conductivity that would be expected to occur would result from the combination of sodicity reductions and the electrolyte effects of gypsum in solution.

The original intention for the experiments with mixed cation solutions of varied salinity was to use the results to determine the influence of salinity and to calculate constants for comparison with the results of McNeal (1968). Calculation of these constants requires equilibration with mixed cations solutions of at least two salt concentrations with subsequent measurement of hydraulic conductivity. However, unexpected trends in hydraulic conductivity were measured when mixed cation solutions were applied. The trends consisted of a sharp increase in hydraulic conductivity followed by a gradual decline which took several months to reach a stable level. These trends were unlike any results found in the literature.

The initial increase in hydraulic conductivity can be partially explained using the leachate chemistry results. The SAR of the leachate solution decreases during the first two weeks of leaching and the resulting decrease in the sodicity of the soil would be expected to cause an increase in hydraulic conductivity. During this same period, the pH also decreased.
Decreases in pH have been associated with reduced clay dispersion and hence increased hydraulic conductivity (Suarez, Rhoades et al. 1984). In addition, the high initial EC of the leachate solutions suggests that salt dissolution has taken place and this may contribute to the electrolyte effect of the applied solution which is expected to cause increases in hydraulic conductivity (McNeal and Coleman 1966).

After hydraulic conductivity reached a maximum value, there was a very gradual decline in hydraulic conductivity and this is more difficult to explain. The soil seems to be in equilibrium with the applied solution as the leachate chemistry remains fairly constant. The decline in K is very slow so whatever chemical or physical process is occurring must only be taking place slowly as well e.g. diffusive processes. In the literature, a number of possible causes for reductions in hydraulic conductivity during column leaching experiments have been identified including, clay dispersion in the presence of low electrolyte concentrations (Pupisky and Shainberg 1979), clay swelling (Mace and Amrhein 2001; McNeal and Coleman 1966), migration of particles leading to pore clogging (Dikinya, Hinz et al. 2008) and biological activity (Klute and Dirksen 1986). However, it is unlikely that the decline is simply associated with a long period of leaching because similar declines did not take place when columns were leached with gypsum.

The solutions commonly used in published studies contained only Na and Ca cations (Frenkel, Goertzen et al. 1978; Keren and Singer 1988; McNeal and Coleman 1966; Shainberg and Caiserman 1971). The presence of Mg in the solutions used in this study may be the cause of the unexpected behaviour. While there is a lack of similar published results for leaching studies, Whitehouse and McCarter (1956) measured changes in clay mineralogy, in the form of diagenic modification, after long periods of saturation with saline waters. These changes were found to occur when montmorillonite clays were saturated with saline waters containing magnesium. The montmorillonite gradually changed to chlorite and illite, and magnesium enriched montmorillonite was the intermediate stage. These changes took several years to occur. It is possible that the gradual decreases in hydraulic conductivity in this study could be the result of changes in clay mineralogy with continued leaching with saline waters. Measurements of the clay mineralogy of the soil used in this study indicate that changes in clay mineralogy may have taken place with the proportion of smectite and illite lower than expected, and the
proportion of kaolinite higher than expected (Table 3-8). However, it is unknown whether these changes can occur during a short leaching period of only a few months.

In the field, long periods of saturation with saline water are not expected to occur, other than on a geological timescale where seawater inundates coastal and other sediments. “Shandying”, the additions of saline groundwaters to the river water used for irrigation would increase the salinity of the irrigation water and according to the results of these experiments, this could significantly increase hydraulic conductivity. If a large proportion of saline groundwater was used and the salinity of the irrigation water was increased to the level of the 100 meq/L as used in these experiments, the magnitude of the increase in hydraulic conductivity could be similar to the maximum effects expected with gypsum applications. However, such high salinities would not normally be considered suitable for use for irrigation. In addition, the electrolyte effects that cause increases in hydraulic conductivity when saline waters are applied to sodic soils are only temporary as the soil sodicity remains high. As soon as low salinity waters such as rainfall infiltrate the soil, the hydraulic conductivity would be reduced again.

There are a number of factors that could contribute to errors in the measurements, including measurement errors and evaporation of collected leachate solutions. The net result of these errors is expected to be an underestimation of the final hydraulic conductivity. As the errors are expected to be fairly consistent across all experimental treatments, the comparisons between treatments and the trends observed would still be valid.

3.6 Conclusions

As the influence of soil chemistry on soil hydraulic properties is soil specific, the magnitude of the changes that would be expected to occur with different treatments could not be predicted. Experiments were therefore conducted to determine the impacts of applying gypsum and mixed cation solutions of varying salinity on hydraulic conductivity.
These experiments included extended periods of soil leaching to gain a better understanding of likely long term impacts on soil hydraulic conductivity.

Application of saturated gypsum solutions for several weeks enabled the maximum possible effects of gypsum on hydraulic conductivity to be determined. The final hydraulic conductivities measured when saturated gypsum solutions were applied were at least ten times higher than the hydraulic conductivities measured using a low salinity mixed cation solution as a surrogate for irrigation water. When saturated gypsum solutions were applied, the hydraulic conductivity gradually increased to a maximum during a leaching period of at least three weeks. The increase in hydraulic conductivity was associated with a substantial reduction in soil sodicity, from an ESP of > 15 to an ESP of < 1.5. At this point, the hydraulic conductivity stabilised.

The columns packed at 1.3 g/cm$^3$ exhibited much higher hydraulic conductivities than those packed at 1.4 g/cm$^3$, when gypsum was applied. For each bulk density, there was significant variation in hydraulic conductivities between replicates. For all of the columns that had gypsum applied, higher final hydraulic conductivities correlated with lower exchangeable magnesium concentrations. This indicates that for these soils, calcium increases hydraulic conductivity more than magnesium.

For the both bulk densities, there was some variation between replicates when time was used as the independent variable. However, when pore volumes were used as the independent variable, the variability between replicates was reduced, typically resulting in one response function. This change suggests that pore size distribution and pore accessibility are key factors influencing the variability in soil chemistry and hydraulic conductivity between replicate columns.

Increases in hydraulic conductivity of the magnitude measured in these experiments could result in increased groundwater recharge but there are a number of factors that would need to be considered when extrapolating results to groundwater management in the Lower Burdekin. The results presented in this study relate to the maximum increase in hydraulic conductivity possible as a result of gypsum applications but this is likely to require high gypsum application rates for extended periods of time. The influence of gypsum on
hydraulic conductivity in the field would be modified by the sporadic application of gypsum and the influence of irrigation water and rainwater composition. In addition, for gypsum applications to lead to increases in groundwater recharge rates, the changes in hydraulic conductivity would need to extend deep into the soil profile to increase connectivity with the groundwater system.

The trends in hydraulic conductivity when 50 meq and 100 meq mixed cation salt solutions were applied consisted of an increase to the maximum hydraulic conductivity during the first 2 to 4 weeks, followed by a gradual decrease in hydraulic conductivity before the hydraulic conductivity stabilised. The maximum hydraulic conductivities for the 100 meq salt solution were approximately double the maximum hydraulic conductivity for the 50 meq salt solution. The time taken to reach the final stable hydraulic conductivity was longer for the 100 meq solution than for the 50 meq solution but the number of pore volumes was almost the same.

The observed trends of a sharp increase in hydraulic conductivity followed by a slow decrease in hydraulic conductivity were unlike results presented in the literature. There are a number of possible reasons for the initial increase in hydraulic conductivity, including decreases in soil sodicity, decreases in pH and enhanced electrolyte effects due to salt dissolution. The reasons for the gradual decline in hydraulic conductivity are less clear. A theory was presented that the clay minerals are undergoing diagenesis due to long periods of saturation with the saline mixed cation solutions. It is not yet known whether these changes can occur during a leaching period of a few months so further investigations are required to assess this.

Based on the results of these experiments, increasing the salinity of the irrigation water applied could significantly increase the hydraulic conductivity of the sodic soils in the BHWSS. The impacts on hydraulic conductivity would be expected to be temporary as the soil sodicity would remain high and any rainfall would decrease the electrolyte effects. The slow decline in hydraulic conductivity after a maximum has been reached is not expected to occur in the field as long periods of saturation with saline waters would not normally to occur.
CHAPTER 4 – APPLICATION OF HYDRUS 1-D TO MODELLING THE RESPONSE OF SODIC SOILS TO AMELIORATION

4.1 Introduction

In Chapter 3, experiments on short columns of sodic clays from the BHWSS demonstrated the impacts of gypsum applications and increased irrigation water salinity on soil chemistry and hydraulic conductivity. In order to assess the potential for sodic soil amelioration to lead to increased deep drainage in the BHWSS, the vertical scale addressed had to be extended from the short columns used in the laboratory towards a scale more pertinent to the investigation of deep drainage. As laboratory experiments at this vertical scale were not possible within the scope of this PhD, a numerical modelling approach was required.

A suitable soil water transport and reaction model was needed to incorporate the key processes that occur when sodic soils are ameliorated. These processes include cation exchange, mineral dissolution and/or precipitation and changes in soil hydraulic conductivity in response to changes in soil chemistry. There are a number of models that can be used to simulate solute transport and reactions but only the UNSATCHEM and HYDRUS-1D models include a function that simulates the influence of soil chemistry on hydraulic conductivity.

Simunek and Suarez (1997) applied the UNSATCHEM model to evaluate a number of sodic soil management strategies. The simulations presented in the Simunek and Suarez (1997) paper comprise of continuous application of a range of treatment solutions, including a saturated gypsum solution.
In the present Chapter, HYDRUS-1D is used to model the experiments of Chapter 3 in order to assess its applicability to simulate sodic soil amelioration at larger scales. As the simulations of Simunek and Suarez (1997) include key features of the laboratory experiments described in Chapter 3, including gypsum applications, continuous application of treatment solutions and assessment of changes in hydraulic conductivity with changes in soil chemistry, these simulations were used as a starting point for developing simulations of the laboratory experiments.

### 4.2 The HYDRUS-1D Modelling Code

The predecessor for the HYDRUS-1D package was HYDRUS version 6.0 (Simunek, Huang et al. 1998), a modelling package capable of simulating variably saturated water flow and solute transport in porous media. Since HYDRUS-1D was first developed, it has been regularly updated and it is now widely used. Version 3.0 of HYDRUS-1D represented a major upgrade to the HYDRUS-1D with the addition of a whole range of additional features. One of these features was the incorporation of the carbonate chemistry module and the carbon dioxide transport and production modules from the UNSATCHEM (Simunek, Suarez et al. 1996) and UNSATCHEM-2D (Simunek and Suarez 1993).

The UNSATCHEM model has been applied previously to simulate sodic soil amelioration (Simunek and Suarez 1997; Suarez 2001). Simunek and Suarez (1997) highlighted the importance of considering the influence of soil chemistry on hydraulic conductivity when predicting the time needed for sodic soil reclamation. In the study by Suarez (2001), the UNSATCHEM model was used in a field study to simulate the reclamation of a saline, sodic soil and predict spatial variations in soil salinity and sodicity.

In this section, the components of the HYDRUS-1D model that are applicable for simulating sodic soil amelioration, including those derived from the UNSATCHEM and UNSATCHEM-2D models, are described.
4.2.1 Variably Saturated Water Flow

The form of the 1-D Richard’s equation that is solved in HYDRUS-1D by a Garlerkin-type linear finite element scheme is shown in equation 4.1

\[
\frac{\partial \theta}{\partial t} = \frac{\partial}{\partial z} \left[ K \left( \frac{\partial h}{\partial z} \right) + \cos \phi \right] - S
\]

(4.1)

where \( h \) is the water pressure head [L], \( \theta \) is the volumetric water content \([L^3/L^3]\), \( t \) is time [T], \( z \) is the spatial coordinate [L], \( S \) is the sink term \([L^3/L^3T^{-1}]\), \( \phi \) is the angle between the flow direction and the vertical axis, and \( K \) is the unsaturated hydraulic conductivity \([LT^{-1}]\).

This form of the Richards equation assumes that the air phase plays an insignificant role in the liquid flow process and that water flow due to thermal gradients can be neglected (Simunek, van Genuchten et al. 2005).

The model allows the user to choose from a range of hydraulic property models including Brooks and Corey (1964) and van Genuchten (1980). The functions in HYDRUS for the van Genuchten (1980) model are as follows:

for \( h < 0 \)

\[
\theta(h) = \theta_s + \frac{\theta_r - \theta_s}{[1 + \left( \frac{\xi h}{\alpha} \right)^{n}]}^m
\]

(4.2)

for \( h \geq 0 \)

\[
\theta(h) = \theta_s
\]

(4.3)

And:

\[
K(h) = K_s \left[ 1 - (1 - S_e^{(im)})^m \right]^2
\]

(4.4)

\[
S_e = \frac{\theta - \theta_i}{\theta_s - \theta_i}
\]

(4.5)

where

\( \theta_s \) = saturated water content [-]

\( \theta_i \) = residual water content [-]
\( \alpha, m, n = \text{empirical parameters} \ [1/L], [-], [-] \)

\( l = \text{pore connectivity parameter} \ [-] \)

\( S_e = \text{effective water content} \ [-] \)

\( K_s = \text{saturated hydraulic conductivity} \ [L/T] \)

Small changes in the shape of the soil water retention curve near saturation can significantly affect the results of numerical simulations of variably saturated flow, including the performance of the numerical scheme itself in terms of stability and rate of convergence (Vogel 2001). There is therefore a need for an alternative to the van Genuchten (1980) model to reduce the non-linearity of the hydraulic conductivity function near saturation. In HYDRUS, this alternative model is based on the Vogel and Cislerova (1988) model and is labelled as the “air-entry option” (equations 4.6 to 4.13). Application of this model alternative is recommended for fine soils (Simunek, van Genuchten et al. 2005).

Use of this alternative model has been shown to effectively remove convergence and oscillation problems (Brooks and Corey 1964). However, predicted infiltration rates are also modified when compared with simulations using the original van Genuchten model (Brooks and Corey 1964). There is therefore a need for further validation of this model for fine-textured soils based on very precise measurements of the unsaturated hydraulic conductivity function close to saturation (Vogel 2001).

For \( h < h_s \)

\[
\theta(h) = \theta_a + \frac{\theta_m - \theta_a}{[1 + (\alpha h)^n]^m}
\]

(4.6)

For \( h \geq h_s \)

\[
\theta(h) = \theta_s
\]

(4.7)

And:

For \( h \leq h_k \)

\[
K(h) = K_s \cdot K_r(h)
\]

(4.8)
for $h_k < h < h_s$

$$K(h) = K_k + \frac{(h - h_k)(K_s - K_k)}{h_s - h_k} \quad (4.9)$$

for $h \geq h_s$

$$K(h) = K_s \quad (4.10)$$

where

$$K_r = \frac{K_k}{K_s} \left( \frac{S_v}{S_{vk}} \right) \left[ F(\theta) - F(\theta_k) \right] \quad (4.11)$$

$$F(\theta) = \left[ 1 - \left( \frac{\theta - \theta_a}{\theta_a - \theta_r} \right)^{1/m} \right]^m \quad (4.12)$$

$$S_{vk} = \frac{\theta_k - \theta}{\theta_k - \theta_k} \quad \theta_m = \text{extrapolated parameter (slightly larger than } \theta_s) \ [-]$$

$$\theta_a = \text{extrapolated parameter (less than or equal to } \theta_r) \ [-]$$

$$\theta_k = \text{extrapolated parameter (less than or equal to } \theta_s) \ [-]$$

$$K_k = K(\theta_k)$$

### 4.2.2 Solute Transport and Reactions

Advective-dispersive chemical transport under transient flow conditions in a partially saturated soil is expressed by (Suarez and Simunek 1996):

$$\frac{\partial \theta c_{T_i}}{\partial t} + \rho \frac{\partial c_{T_i}}{\partial t} + \rho \frac{\partial \tilde{c}_{T_i}}{\partial t} = \frac{\partial}{\partial z} \left[ \theta D \frac{\partial c_{T_i}}{\partial z} - q c_{T_i} \right] \quad (4.14)$$

$i = 1, n_s$
where

- $c_{Ti}$ = the total dissolved concentration of the aqueous component $i$ [ML$^{-3}$]
- $\bar{c}_{Ti}$ = the total exchangeable concentration of the aqueous component $i$ [MM$^{-1}$]
- $\hat{c}_{Ti}$ = the non-adsorbed solid phase concentration of aqueous component $i$ [MM$^{-1}$]
- $\rho$ = soil bulk density [ML$^{-3}$]
- $D$ = dispersion coefficient [L$^2$T$^{-1}$]
- $q$ = volumetric flux [LT$^{-1}$]
- $n_s$ = the number of components

The major ion chemistry module in HYDRUS is applicable for modelling agricultural scenarios because the impacts of evapotranspiration, cation exchange, salt accumulation and dissolution of common minerals (including gypsum and calcite) on root zone chemistry can be simulated. This module enables the simulation of equilibrium reactions involving Ca, Mg, Na, K, SO$_4$, Cl, NO$_3$, H$_4$SiO$_4$, HCO$_3$ and CO$_2$. The equilibrium equations are solved using an iterative approach and the phases considered are aqueous, gaseous, adsorbed, precipitated and complex. The soil solution pH is determined based on the alkalinity equation (equation 4.15) and a charge balance expression. Alkalinity is modelled as a conservative property to allow for fluctuation of carbon dioxide concentrations (Simunek, van Genuchten et al. 2005).

\[
\text{Alkalinity} = 2\text{CO}_3^\text{T} + \text{HCO}_3^\text{T} + [\text{OH}^-] - [\text{H}^+] \quad (4.15)
\]

(where the “T” denotes the total analytical concentration in solution of carbonate and bicarbonate)

### 4.2.3 Chemistry Dependent Hydraulic Conductivity

The major ion chemistry module in HYDRUS also includes a function which accounts for the impact of solution chemistry on hydraulic conductivity. This function is based on the work of McNeal (1968) and Suarez, Rhoades et al. (1984). The effects of soil chemistry on hydraulic conductivity appear to be soil specific but by using this function, qualitative predictions of these effects can be made. The function is incorporated into HYDRUS as a scaling parameter $r$ that is dependent on the salt concentration of the solution, the
exchangeable sodium percentage of the soil and the pH of the soil solution. The value of \( r \) equals 1 where the solution chemistry enables a maximum hydraulic conductivity. The scaling parameter \( r \) is thus included in the hydraulic conductivity function:

\[
K(h) = rK_sK_r = rK_sS_e^{1/2}[1-(1-S_e^{1/m})m]^2
\]  

(4.16)

The scaling parameter is divided into two parts as follows:

\[
r(pH, SAR, C_0) = r_1(SAR, C_0)r_2(pH)
\]

(4.17)

where \( r_1 \) represents the effect of ESP (SAR) and soil solution salinity, \( C_0 \), on the hydraulic conductivity and \( r_2 \) accounts for the effect of the soil solution pH on hydraulic conductivity.

### 4.2.4 Additional features in HYDRUS 1-D

Some additional features of HYDRUS which are applicable for modelling soils under irrigation include the incorporation of root uptake functions with a database of suggested values for root uptake parameters (Nolan, Randall Bayless et al. 2005) and the choice of boundary conditions. Model users can choose to include an “atmospheric” upper boundary condition which can be used to specify rainfall, irrigation and evaporation or evapotranspiration on a daily basis (Schoups, Hopmans et al. 2006). Lower boundary conditions that can be applied for simulating irrigated soils include a constant zero pressure head at a specific depth, below the root zone, to represent cases with shallow groundwater (Schoups, Hopmans et al. 2006) and a free lower drainage boundary condition for cases where the groundwater tables are deep (Jimenez-Martinez, Skaggs et al. 2009; Schoups, Hopmans et al. 2006). HYDRUS-1D has recently been applied for estimation of groundwater recharge under irrigated land using this free drainage lower boundary condition (Jimenez-Martinez, Skaggs et al. 2009).
4.3 Application of HYDRUS 1D to Simunek and Suarez (1997) Scenarios

Simunek and Suarez (1997) used the UNSATCHEM model to demonstrate the application of the hydraulic conductivity reduction function (equation 4.16) and to assess a range of sodic soil reclamation strategies for a loam soil. In this Chapter, HYDRUS-1D version 4.09 (http://www.pc-progress.cz/) was applied to the published Simunek and Suarez (1997) UNSATCHEM models to determine whether HYDRUS-1D version 4.09 was able to reproduce these earlier results. The HYDRUS-1D model has many additional features when compared to UNSATCHEM. While the major ion chemistry module was adapted from the UNSATCHEM (Simunek, Suarez et al. 1996) and UNSATCHEM-2D (Simunek and Suarez 1993) models, the flow routines in HYDRUS-1D are different from those in UNSATCHEM (Simunek 2009). In light of differences between UNSATCHEM and HYDRUS-1D version 4.09, it was deemed worthwhile to assess the applicability of HYDRUS-1D to modelling sodic soil amelioration using a comparison between published UNSATCHEM simulations of sodic soil amelioration and simulations of the same cases using the HYDRUS-1D model.

The HYDRUS-1D simulations in this Chapter were initially set up using the input data provided in the published paper (Simunek and Suarez 1997). The information that was provided in the paper consists of geometry information, soil hydraulic properties, solution composition for the different treatments and some of the solute transport and reaction parameters. Preliminary tests showed that the additional parameters that were needed to reproduce the published results were the molecular diffusion coefficient, the dispersivity, the Ca/K cation exchange coefficient, the initial carbon dioxide concentrations and the carbon dioxide production parameters. Subsequently, the original UNSATCHEM input files for the published paper were provided by the author (Simunek 2008) and these files were used to set up models in HYDRUS-1D version 4.09. These model inputs are summarized in a table in the appendix of this thesis.

The model inputs presented in the subsequent sections are exactly the same as those used in the published simulations.
4.3.1 Soil Hydraulic Properties

The hydraulic property model used was the van Genuchten (1980) model and the soil hydraulic properties are given in Table 4-1. These properties represent a loam soil. The hydraulic conductivity noticeably increases towards $K_s$ once the pressure head increases above -200cm (Figure 4-1).

Table 4-1. Soil hydraulic properties for loam soil used in Simunek and Suarez 1997 model.

<table>
<thead>
<tr>
<th>$\theta_r$</th>
<th>$\theta_s$</th>
<th>$\alpha$ (cm$^{-1}$)</th>
<th>$n$</th>
<th>$K_s$ (cm/day)</th>
<th>$l$</th>
</tr>
</thead>
<tbody>
<tr>
<td>0</td>
<td>0.48</td>
<td>0.015</td>
<td>1.592</td>
<td>60.48</td>
<td>0.5</td>
</tr>
</tbody>
</table>

Figure 4-1. Hydraulic conductivity as a function of hydraulic head.

The hydraulic conductivity depends on soil chemistry as described by the hydraulic conductivity reduction function (equation 4.16). This equation reduces hydraulic conductivity when the soil chemistry is below “optimum conditions” to account for the effects of soil swelling. The “optimum conditions” with reference to the hydraulic conductivity function in HYDRUS-1D equate to a low pH, below 6.9, and a high salt concentration, at least 300 meq/L. The simulations were run with and without using the hydraulic conductivity reduction function. The reason for this was to show the differences in predicted timing for reclamation with and without consideration of the influence of chemistry on hydraulic properties and thus demonstrate the importance of considering hydraulic conductivity responses to changes in chemistry.
4.3.2 Discretization

The height of the soil profile being modelled was 100 cm. 51 nodes were used to discretize the profile, giving 2 cm spaced grid cells as used by Simunek and Suarez (1997). The grid cells were evenly distributed.

The final time for the simulation varied between the reclamation options, i.e. gypsum in irrigation water, gypsum in soil, acidified water and high quality water, due to differences in the time needed for reclamation. The maximum simulation time for the models was 238 years. The initial time step was 0.001 days and the maximum time step was 5 days. The maximum number of iterations per time step was 20.

4.3.3 Initial Conditions

The soil was modelled as an initially dry soil that was saturated at the surface. The initial pressure head condition was -500 cm down the profile and 0 cm at the soil surface.

The initial concentrations of exchangeable cations were 40 meq/kg for Ca and Mg and 120 meq/kg for Na. The initial soil solution composition was dominated by Na and Cl ions, with 4.8 meq/L of each. The soil solution also contained 0.2 meq/L Ca, 0.2 meq/L Mg and 0.4 meq/L alkalinity. The units were listed as mmol in the paper but as meq in the model inputs.

4.3.4 Boundary Conditions

A constant value of \( h = 0 \) was maintained at uppermost grid cell, \( z = 0 \). The lower boundary condition chosen is “free drainage”, a unit hydraulic gradient boundary condition. This boundary condition allows drainage to occur once the soil profile is saturated. The drainage rate assigned to the bottom node is equal to \( K(h) \), where \( h \) is the local value of the pressure head and \( K(h) \) is the hydraulic conductivity corresponding to this pressure head.
The upper boundary condition used for solute transport was a Cauchy type boundary condition used to prescribe the concentration flux represented by the applied solution in each case (Equation 4.18). This boundary condition has been shown to conserve mass whereas the alternative, a Dirichlet type boundary condition where concentration is prescribed at the boundary, has been found to overestimate solute concentrations (Leij, Skaggs et al. 1991). The lower boundary condition was a zero concentration gradient, a Neumann type boundary condition (Equation 4.19). This boundary condition is used when water flow is directed out of the region (Simunek, van Genuchten et al. 2005).

\[- \theta D \frac{\partial c}{\partial x} + qc = q_0 c_0 \quad \text{at } x = 0 \quad (4.18)\]
\[\theta D \frac{\partial c}{\partial x} = 0 \quad \text{at } x = L \quad (4.19)\]

### 4.3.5 Solute Transport Parameters

The solute transport parameters used were a bulk density of 1.3 g/cm$^3$, a molecular diffusion coefficient in free water of 30 cm$^2$/day and a longitudinal dispersivity of 0 cm. In the model, a dispersion coefficient is used to represent the combined effects of dispersion and molecular diffusion (Equation 4.20)

\[D = \alpha \frac{|q|}{\theta} + D_m \tau \quad (4.20)\]

where $\alpha$ is dispersivity [L], $\theta$ is the water content [L$^3$L$^{-3}$], $|q|$ is the Darcy water flux [LT$^{-1}$], $D_m$ is the molecular diffusion coefficient [L$^2$T$^{-1}$], and $\tau$ is the tortuosity (dimensionless) (Schoups, Hopmans et al. 2006). The tortuosity is calculated as a function of moisture content as follows (Millington and Quirk 1961) (Equation 4.21):

\[\tau = \frac{\theta^{7/3}}{\theta_s} \quad (4.21)\]

(where $\theta_s$ is the saturated moisture content)
Equilibrium equations are used to represent cation exchange, i.e. exchange between the exchangeable cations on the clay surfaces and cations in solution. The Gapon form of the exchange equation (equation 4.22) has been used widely for investigating Na-Ca exchange and the reclamation of sodic soils. However, this form is not consistent with thermodynamics of exchange reactions whereas the Vanselow form is (Evangelou and Phillips 1988). The Gapon exchange selectivity coefficient has previously been shown to be in close agreement with the Vanselow exchange selectivity coefficient but experimental results have shown that this is only the case for Vanselow coefficients <0.015 and sodium equivalent fractions in the exchange of <0.20 (Evangelou and Phillips 1987).

\[
K_{ij} = \frac{c_i^{y+}}{c_j^{x+}} \left( \frac{c_j^{x+}}{c_i^{y+}} \right)^{1/y} \left( \frac{c_i^{1/y}}{c_j^{1/y}} \right)^{1/x} \tag{4.22}
\]

where \( K_{ij} \) is the Gapon selectivity coefficient, \( x \) and \( y \) are the valencies of species \( i \) and \( j \) and the overscored concentrations are those of the exchanger phase (in mol./kg soil).

The Gapon form is used to represent cation exchange between specific combinations of cations and the parameters needed in the model are the cation exchange coefficients. Equations 4.23 and 4.24 show the cation exchange coefficients, \( K_{ij} \) for Mg/Ca and Ca/Na exchange respectively, where the concentrations of adsorbed species are expressed in moles of charge / kg and the ions in solution are expressed as ion activities (Simunek, van Genuchten et al. 2005). Cation exchange coefficients are needed to describe exchange between the four major cations; calcium, magnesium, sodium and potassium. The cation exchange capacity (CEC) is equal to the sum of concentrations of these four cations.

\[
K_{13} = \frac{Mg^{2+}}{Ca^{2+}} \left( Ca^{2+} \right)^{1/2} \tag{4.23}
\]

\[
K_{14} = \frac{Ca^{2+}}{Na^+} \left( Na^+ \right)^{1/2} \tag{4.24}
\]
Selectivity coefficients are often assumed to be constant for a given soil but there is evidence that these coefficients can vary with both ionic strength and clay mineralogy (Kopittke, So et al. 2006). In HYDRUS-1D, both the CEC and the selectivity coefficients are assumed to be independent of solution conditions and as such they remain constant. This simplification would often be necessary as variation in CEC and selectivity coefficients in response to changes in solution conditions are not normally measured. For the Simunek and Suarez (1997) replication models in this Chapter, the CEC of the soil was specified as 200 meq/kg and the Gapon selectivity coefficients used were 0.896 for Ca/Mg, 1.158 for Ca/Na and 0.111 for Ca/K.

4.4.8 Reclamation Options

Simunek and Suarez (1997) simulated seven reclamation strategies in total. These reclamation strategies were chosen to cover the most commonly recommended strategies from the literature. A summary of these strategies is provided by Oster and Jayawardane (1997) who state that gypsum is the calcium source most commonly used to reclaim sodic soils and that suitable amendments for calcareous, i.e. containing calcite, sodic soils include sulphuric acid and acid-forming materials. The acids or acid forming materials react with the calcite and provide a soluble source of calcium within the soil. Where there are elevated levels of carbon dioxide in the soil atmosphere, the solubility of calcite is increased even when irrigation water alone is used for reclamation (Oster and Jayawardane 1997). The reclamation strategies simulated by Simunek and Suarez (1997) therefore included gypsum treatments, both in irrigation water and incorporated into the soil, acidified water with calcite in the profile and irrigation water with calcite in the profile Table 4-2. The compositions of the applied solutions is given in Table 4-3.
Table 4-2. Summary of the reclamation scenarios simulated by Simunek and Suarez 1997.

<table>
<thead>
<tr>
<th>Applied Solution</th>
<th>Precipitated Minerals</th>
<th>Carbon Dioxide Model</th>
</tr>
</thead>
<tbody>
<tr>
<td>High quality water</td>
<td>None</td>
<td>Model-predicted CO₂</td>
</tr>
<tr>
<td>Gypsum saturated water</td>
<td>None</td>
<td>Constant CO₂</td>
</tr>
<tr>
<td>High quality water</td>
<td>Gypsum incorporated into the top 20 cm</td>
<td>Constant CO₂</td>
</tr>
<tr>
<td>High quality water</td>
<td>Calcite throughout the profile</td>
<td>Constant CO₂</td>
</tr>
<tr>
<td>High quality water</td>
<td>Calcite throughout the profile</td>
<td>Model-predicted CO₂</td>
</tr>
<tr>
<td>Acidified water at pH 2.05</td>
<td>Calcite throughout the profile</td>
<td>Constant and model-predicted CO₂</td>
</tr>
<tr>
<td>Acidified water at pH 1.09</td>
<td>Calcite throughout the profile</td>
<td>Constant and model-predicted CO₂</td>
</tr>
</tbody>
</table>

Table 4-3. Compositions of applied solutions, in meq/L, for the Simunek and Suarez 1997 models.

<table>
<thead>
<tr>
<th></th>
<th>Ca</th>
<th>Mg</th>
<th>Na</th>
<th>Alkalinity</th>
<th>SO₄</th>
<th>Cl</th>
</tr>
</thead>
<tbody>
<tr>
<td>High quality water</td>
<td>1.5</td>
<td>0.5</td>
<td>2.0</td>
<td>0.5</td>
<td>2.5</td>
<td>1.0</td>
</tr>
<tr>
<td>Gypsum saturated water</td>
<td>32.0</td>
<td>0.5</td>
<td>2.0</td>
<td>0.5</td>
<td>33.0</td>
<td>1.0</td>
</tr>
<tr>
<td>Acidified water 1</td>
<td>1.5</td>
<td>0.5</td>
<td>2.0</td>
<td>-10.0</td>
<td>11.0</td>
<td>3.0</td>
</tr>
<tr>
<td>Acidified water 2</td>
<td>1.5</td>
<td>0.5</td>
<td>2.0</td>
<td>-100.0</td>
<td>11.0</td>
<td>93.0</td>
</tr>
</tbody>
</table>

4.4 Results

In this section we compare our simulation results with those of Simunek and Suarez (1997). The published paper included 27 graphs of the model outputs. These graphs covered a range of reclamation strategies in which the majority of simulations included the “hydraulic conductivity reduction due to chemistry” option (Table 4-4).
Table 4-4. Reclamation strategies simulated and outputs presented in Simunek and Suarez (1997) paper.

<table>
<thead>
<tr>
<th>Reclamation Strategy</th>
<th>Outputs Presented</th>
</tr>
</thead>
<tbody>
<tr>
<td>Irrigation with high-quality water and no amendment</td>
<td>Pressure head profiles, hydraulic conductivity profiles and sodium adsorption ratio (with and without the “hydraulic conductivity reduction due to chemistry” option)</td>
</tr>
<tr>
<td>Irrigation with gypsum saturated water</td>
<td>Hydraulic conductivity and sodium adsorption ratio (with and without the “hydraulic conductivity reduction due to chemistry” option)</td>
</tr>
<tr>
<td>Irrigation with high-quality water and gypsum incorporated into the top 20 cm</td>
<td>Hydraulic conductivity and infiltration rate (with the “hydraulic conductivity reduction due to chemistry” option), sodium adsorption ratio (with and without the “hydraulic conductivity reduction due to chemistry” option)</td>
</tr>
<tr>
<td>Irrigation with high-quality water, calcite throughout the profile and constant CO₂</td>
<td>Hydraulic conductivity, infiltration rate and sodium adsorption ratio (with the “hydraulic conductivity reduction due to chemistry” option)</td>
</tr>
<tr>
<td>Irrigation with high-quality water, calcite throughout the profile and model-predicted CO₂</td>
<td>Hydraulic conductivity, infiltration rate, sodium adsorption ratio, CO₂ and alkalinity (with the “hydraulic conductivity reduction due to chemistry” option)</td>
</tr>
<tr>
<td>Infiltration of acid water at pH 2.05 into calcareous soil</td>
<td>Sodium adsorption ratio profiles with constant CO₂ and model predicted CO₂ (with the “hydraulic conductivity reduction due to chemistry” option)</td>
</tr>
<tr>
<td>Infiltration of acid water at pH 1.09 into calcareous soil</td>
<td>Sodium adsorption ratio profiles with constant CO₂ and model predicted CO₂ and calcite profiles (with the “hydraulic conductivity reduction due to chemistry” option)</td>
</tr>
</tbody>
</table>

The most common simulation results presented were the sodium adsorption ratio (SAR) and the hydraulic conductivity. The SAR is included as a measure of sodicity and therefore of sodic soil reclamation. The sodium in the soil solution is assumed to be in equilibrium with the soil exchangeable sodium.
Use of the hydraulic conductivity reduction due to chemistry option causes the hydraulic conductivity to be reduced to below the saturated hydraulic conductivity when the chemical conditions are not optimum. When the hydraulic conductivity is reduced the rate of flow of the applied solution through the soil is slower. This reduction in flow rate also slows down changes in soil chemistry. The results from all of the reclamation scenarios showed an increase in the proportion of the profile with a higher hydraulic conductivity corresponding with a decrease in the sodium content of the soil.

4.4.1 High Quality Water Reclamation Results

The results for high quality water were used to emphasize the changes in soil chemistry and soil hydraulic conductivity profiles with and without the “hydraulic conductivity reduction due to chemistry” option. When the “hydraulic conductivity reduction due to chemistry” is not turned on, the soil profile becomes saturated and the hydraulic conductivity reaches the saturated hydraulic conductivity ($K_s$) within 0.5 days. In contrast, when the “hydraulic conductivity reduction due to chemistry” is turned on, the hydraulic conductivity is reduced due to the initial sodicity of the soil and the composition of the applied water. As a result, the soil profile takes much longer to saturate i.e. 1 year.

The SAR results illustrate how the cation composition of the soil solution is altered as a result of exchange reactions between the adsorbed cations and the cations in the applied solution. For the scenario where high quality water was applied and the “hydraulic conductivity reduction due to chemistry” option was not used, the SAR of the soil solution was originally quite high due to the sodic water initially in the soil and the adsorbed sodium initially present in the soil. The SAR is gradually reduced down the profile towards the low SAR of the applied solution. When “hydraulic conductivity reduction due to chemistry” option was used, the SAR of the soil solution remained higher for longer, partly due to the increased time needed to saturate the soil profile.

For ease of comparison, the published results are provided in the left hand side of Figures 4-2 to 4-12, with the corresponding HYDRUS results on the right hand side. For the models with high quality water, the HYDRUS results where the “hydraulic conductivity
reduction due to chemistry” option was not used matched the published UNSATCHEM results fairly well (Figure 4-2). When the “hydraulic conductivity reduction due to chemistry” option was used, the HYDRUS results differed from the UNSATCHEM results quite significantly for hydraulic conductivity and SAR (Figure 4-3). The hydraulic conductivity at 200 years was greater than 40 cm/day for the HYDRUS results whereas for the UNSATCHEM results, the hydraulic conductivity at 200 years was less than 20 cm/day. The HYDRUS results for SAR consisted of a much quicker decrease in SAR when compared to the UNSATCHEM results. For the HYDRUS model results, the SAR was reduced to 2 after 200 years whereas for the UNSATCHEM model, the SAR was still higher than 8 after 200 years.
Figure 4-2. Irrigation with high-quality water and no amendment (without “hydraulic conductivity reduction due to chemistry” option).
Figure 4-3. Irrigation with high-quality water and no amendment (with “hydraulic conductivity reduction due to chemistry” option)
4.4.2 Gypsum Reclamation Results

The results for reclamation with a gypsum saturated solution show a reduction in the sodicity as illustrated by the decreases in the sodium adsorption ratio. This corresponded with an increase in the soil hydraulic conductivity over the same time period. When a gypsum saturated solution was applied to the sodic soil and the “hydraulic conductivity reduction due to chemistry” option was not used, the SAR reduced to 0 within 10 days. However, when the effects of hydraulic conductivity reductions due the initially sodic soil are taken into account, it takes 200 days for the SAR to be reduced to 0. For the models with gypsum saturated solutions, the HYDRUS results matched the published UNSATCHEM results reasonably well (Figure 4-4 and Figure 4-5).

The results for models with gypsum incorporated into the top 20 cm of soil were similar to the results for the models with gypsum saturated water. This is to be expected as the processes involved are the same. The main differences between the two groups of model results are the timing for reclamation, which was faster when gypsum was incorporated into the soil, and the shape of the initial SAR profiles, as an abrupt change in SAR at the top of the profile occurred when gypsum was incorporated into the soil.

For the model with gypsum incorporated into the top 20 cm without the “hydraulic conductivity reduction due to chemistry” option, the HYDRUS results matched the published UNSATCHEM results fairly closely (Figure 4-6). For the model with gypsum incorporated into the top 20 cm with the “hydraulic conductivity reduction due to chemistry” option, the HYDRUS results were similar to the published UNSATCHEM results but there were slight differences in the timing (Figure 4-7). The time taken for changes in SAR and hydraulic conductivity was greater for the HYDRUS results.
Figure 4.4. Irrigation with gypsum saturated water (with “hydraulic conductivity reduction due to chemistry” option)
Figure 4-5. Irrigation with gypsum saturated water (without “hydraulic conductivity reduction due to chemistry” option).

Figure 4-6. Irrigation with high-quality water and gypsum incorporated into the top 20 cm (without “hydraulic conductivity reduction due to chemistry” option).
Figure 4-7. Irrigation with high-quality water and gypsum incorporated into the top 20 cm (with "hydraulic conductivity reduction due to chemistry" option).
4.4.3 Results for Reclamations with Calcite in the Soil Profile

For the models with high quality water applied and calcite throughout the profile, the SAR decreased over time and the hydraulic conductivity increased over time but the changes were slower than for the gypsum reclamation models. The published results for these models all included the “hydraulic conductivity reduction due to chemistry” option. For these models the carbon dioxide was either simulated using a constant carbon dioxide concentration or model predicted carbon dioxide. When model predicted carbon dioxide was used, the change in SAR and hydraulic conductivity was more rapid.

In the case where model predicted carbon dioxide was used (Figure 4-8), the HYDRUS results closely matched the published UNSATCHEM results but in the case with constant carbon dioxide (Figure 4-9) the results differed. In this case the hydraulic conductivity and SAR changed more slowly using the HYDRUS model compared with the published UNSATCHEM results.

For the models with acidified water applied and calcite throughout the profile the HYDRUS results for the SAR were similar to the HYDRUS results with calcite and high quality water. The UNSATCHEM results showed that the acidified water and calcite in the profile resulted in a quicker reclamation than the dilute water with calcite in the profile. For all of the models with acidified water, the HYDRUS results showed much slower changes in the SAR than the published UNSATCHEM results (Figures 4.10 to 4.13). In addition, oscillations in the SAR results occurred at 10 days for the pH 2.05 water with model predicted carbon dioxide concentrations (Figure 4-11). It is not known why these oscillations were present for this scenario at 10 days yet were absent for the other time periods.
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Figure 4-8. Irrigation with high quality water and calcite throughout profile, model predicted CO$_2$ (with “hydraulic conductivity reduction due to chemistry” option).
Figure 4-9. Irrigation with high-quality water, calcite throughout the profile and constant CO$_2$ (with “hydraulic conductivity reduction due to chemistry” option).
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Figure 4-10. Infiltration of acid water at pH 2.05 into calcareous soil with constant CO₂ (with “hydraulic conductivity reduction due to chemistry” option).

Figure 4-11. Infiltration of acid water at pH 2.05 into calcareous soil with model predicted CO₂ (with “hydraulic conductivity reduction due to chemistry” option).

Figure 4-12. Infiltration of acid water at pH 1.09 into calcareous soil with constant CO₂ (with “hydraulic conductivity reduction due to chemistry” option).
Figure 4-13. Infiltration of acid water at pH 1.09 into calcareous soil with model predicted CO$_2$ (with “hydraulic conductivity reduction due to chemistry” option).
4.4.4 Summary of Simunek and Suarez (1997) Replication Results

All simulated reclamation strategies resulted in a reduction in the sodicity and an increase in hydraulic conductivity. The difference between the results for the different strategies was in the timing and the maximum hydraulic conductivity that was attained.

Out of all of the HYDRUS model results, the following reclamation options resulted in the poorest matches with the published UNSATCHEM results:

1. high quality water without the “hydraulic conductivity reduction due to chemistry” option (Figure 4-2),
2. high quality water with calcite and constant CO$_2$ (Figure 4-9), and
3. the reclamations with acidified water (Figure 4-10, Figure 4-11, Figure 4-12 and Figure 4-13).

Where there were differences between the two sets of results, they typically related to timing.

In order to determine the reason for the differences between the published UNSATCHEM results and the HYDRUS results, the model files were run in UNSATCHEM and the same series of graphs were produced. These graphs matched the published UNSATCHEM graphs except for a couple of minor differences in six of the graphs which appear to be related to labelling and the smoothness of the infiltration rate curves. As the UNSATCHEM model files and the HYDRUS files had identical model inputs, the differences between the UNSATCHEM results and the HYDRUS results must be due to differences between the model codes.

For the reclamation option with acidified water at pH 2.05, calcite throughout the profile and model predicted CO$_2$, the number of nodes used to discretize the profile was increased in stages to determine the effect on the model outputs. The increases in the number of nodes lead to a closer match between the UNSATCHEM outputs and the HYDRUS outputs for SAR.

Further tests were performed using the reclamation model with gypsum saturated water to determine the impact of a range of factors that affect hydraulic conductivity and ion...
exchange processes. These factors include soil bulk density, longitudinal dispersivity and Gapon selectivity constants. Increasing the bulk density slowed down the changes in hydraulic conductivity. Increasing the dispersivity also slowed down the changes in the SAR and hydraulic conductivity as well as leading to discontinuities in the hydraulic conductivity curves. As the Ca/Na Gapon exchange coefficient was increased the rate at which the SAR changed was quicker and the SAR values were higher.

As the profile discretization, bulk density, dispersivity and cation exchange coefficients were determined to influence hydraulic conductivity profiles and therefore predicted reclamation times, they will need to be carefully considered in further modelling. Sensitivity testing will be used to determine the influence of these parameters when modelling sodic clay soils.

### 4.5 Column Experiment Model Setup

The model setup in the Simunek and Suarez (1997) paper was used as the starting point for set up models to simulate the column experiments in Chapter 3. Where measured data specific to the experiment setup or the soils tested was available, this data was incorporated into the model setup.

The model inputs that were kept the same as those in the results of Simunek and Suarez (1997) paper were the model version, the boundary conditions and the initial soil profile pressure heads. The first model inputs that were altered were the length of the soil profile modelled and the soil type for the hydraulic properties. After these changes were made, further changes were needed to improve model stability. These changes were in addition to the changes in soil chemistry. Sensitivity testing was used to choose some of the parameters that have not been measured.
4.5.1 Soil Hydraulic Properties

The hydraulic property model used was the van Genuchten (1980) model and the soil hydraulic parameters used (Table 4-5) were the in-built parameters for a “clay” soil. However, the saturated hydraulic conductivity was decreased to 2.2 cm/day, the maximum hydraulic conductivity measured in the laboratory experiments described in Chapter 3.

Table 4-5. Soil hydraulic properties for a clay soil, used in column experiment model.

<table>
<thead>
<tr>
<th>$\theta_r$</th>
<th>$\theta_s$</th>
<th>$\alpha$</th>
<th>$n$</th>
<th>$K_s$</th>
<th>1</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.068</td>
<td>0.38</td>
<td>0.008</td>
<td>1.09</td>
<td>2.2</td>
<td>0.5</td>
</tr>
</tbody>
</table>

For the modelling of fine textured soils, the HYDRUS manual (Simunek, van Genuchten et al. 2005) recommends the additions of an air entry value ($h_s = -2$cm). Using this option is believed to improve model stability (Simunek 2009). Choosing this option alters the hydraulic property function to a modified van Genuchten function (Figure 4-14). The resulting hydraulic conductivity versus pressure head function is less steep close to saturation. When the air entry option is not used, there are large changes in hydraulic conductivity in response to small changes in pressure head near saturation. Whereas, when the air entry option is used, the hydraulic conductivity increases more gradually but starts to increase at pressure heads of less than -200cm.
As mentioned earlier, the saturated hydraulic conductivity is altered from the $K_s$ in the input data when the “hydraulic conductivity reduction due to chemistry” option is applied. The simulations of the column experiments were run using this option.

### 4.5.2 Discretization

The depth of the soil profile being modelled was 4 cm as this was the height of the column used in laboratory experiments. The grid cells were evenly distributed. Using a regular grid can decrease numerical truncation errors (Zheng and Bennett 2002). 241 nodes were used to discretize the profile, giving 0.017 cm spaced grid cells, after a grid independence test indicated that such a grid was sufficient. This node spacing is finer than the default grid spacing in HYDRUS and finer than that used for the Simunek and Suarez (1997) paper. Using a sufficiently fine grid spacing helps to ensure a small Peclet number and minimize numerical dispersion (Zheng and Bennett 2002).
The minimum time step and initial time step were reduced to below the default values in order to improve the stability of the numerical solution. When the models were first set up, an initial time step of 0.001 days and minimum time step of $1 \times 10^{-5}$ days were used. It was determined that reducing the initial and minimum time steps reduced mass balance errors and improved model stability. Therefore, the initial time step was reduced to $1 \times 10^{-6}$ days and the minimum time step was reduced to $1 \times 10^{-7}$ days. The maximum time step was 1 day and the maximum number of iterations per time step was 20.

### 4.5.3 Initial Conditions

The soil used in the laboratory experiments had a cation exchange capacity of 290 meq/kg and the initial concentrations of exchangeable cations were 102 meq/kg for Ca, 136 meq/kg for Mg, 49 meq/kg for Na and 3 meq/kg for K. “Burdekin River water” was used as the initial soil solution (Table 4-6) as the soils have been irrigated using this water for a period of at least ten years.

Table 4-6. The composition of “Burdekin River water”, concentrations in meq/L.

<table>
<thead>
<tr>
<th></th>
<th>Ca</th>
<th>Mg</th>
<th>Na</th>
<th>K</th>
<th>Alkalinity</th>
<th>SO₄</th>
<th>Cl</th>
</tr>
</thead>
<tbody>
<tr>
<td>Burdekin River water</td>
<td>0.75</td>
<td>0.64</td>
<td>0.78</td>
<td>0.08</td>
<td>1.66</td>
<td>0.11</td>
<td>0.59</td>
</tr>
</tbody>
</table>

The data for the “Burdekin River water” composition was obtained from the Department of Environment and Resource Management HYDSYS database. The sampling site chosen, “Burdekin River at Clare: 120006B”, is close to the Mulgrave area of the BHWSS. The 2008 analysis results were chosen as a representative composition for the Clare sampling location, 120006B, because these results were consistent with the mean and median values for the long term measurements at this sampling location.

### 4.5.4 Solute Transport Parameters

The molecular diffusion coefficient was maintained at 30 cm$^2$/day, the value used in the Simunek and Suarez (1997) models. Molecular diffusion may be a significant mechanism of transport in cases when flow velocities are low (Zheng and Bennett 2002).
longitudinal dispersivity was increased from 0 to 1 cm. Typical values for dispersivity in packed columns range from 0.1 to 2 cm (Jury and Horton 2004). The bulk density was set at 1.3 g/cm$^3$ because this was the value most commonly used in the experiments. Sensitivity testing was used to determine the influence of the molecular diffusion coefficient, the longitudinal dispersivity and the bulk density on cation exchange and hydraulic conductivity.

The Gapon selectivity coefficients were altered to achieve a close match between the model outputs and the USDA published SAR-ESP relationship. This relationship was assumed when calculating the concentration of cations in the mixed cation solutions for the laboratory experiments in the absence of measured relationships for the local soils. The Gapon selectivity coefficients used were 1 for Ca/Mg, 2 for Ca/Na and 0.111 for Ca/K.

$$ESP = \frac{100(-0.0126 + 0.01475SAR)}{1 + (-0.0126 + 0.01475SAR)}$$  \hspace{1cm} (4.20)

(USDA 1954)

### 4.5.5 Applied Solutions

“Low salinity water”, “gypsum saturated water” and “high salinity water” have the same compositions as the solutions used in the laboratory column experiments described in Chapter 3. The ion concentrations for the applied solutions (in meq/L) are presented in Table 4-7.

<table>
<thead>
<tr>
<th>Solution</th>
<th>Ca</th>
<th>Mg</th>
<th>Na</th>
<th>K</th>
<th>Alkalinity</th>
<th>SO$_4$</th>
<th>Cl</th>
</tr>
</thead>
<tbody>
<tr>
<td>Low salinity solution</td>
<td>0.75</td>
<td>0.64</td>
<td>0.78</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>2.17</td>
</tr>
<tr>
<td>Gypsum saturated water</td>
<td>31.0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>31.0</td>
<td>0</td>
</tr>
<tr>
<td>High salinity water</td>
<td>20.5</td>
<td>50.5</td>
<td>59</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>100</td>
</tr>
</tbody>
</table>
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4.6 Results

The model outputs that were used to determine the model stability included the hydraulic conductivity profiles, the chemical mass balance errors and the water mass balance errors. A number of the model input parameters were found to have a significant impact on these model outputs, these include: the hydraulic property function used, the initial soil solution chemistry, the alkalinity of the applied solution and the number of nodes used to discretize the soil profile.

The overall trends in the results were compared to the column experiment results. In particular, the trends in hydraulic conductivity and exchangeable cation concentrations over time were compared. The timing associated with these trends was also noted but exact matches were not expected as the model was not calibrated in order to match the experimental results.

4.6.1 Hydraulic Property Functions

Choosing the air entry option alters the relationship between hydraulic conductivity and pressure head. For simulations that were unstable, adding the air entry value improved the model stability. A good example of this is with the results where a gypsum solution is applied. When the air entry value is not used, there are instabilities in the hydraulic conductivity profiles (Figure 4-15) whereas, when the air entry value is used, the hydraulic conductivity profiles are smooth (Figure 4-16). For simulations that were already stable, turning off the air entry option did not alter the hydraulic conductivity profiles.

Some of the instabilities shown in the hydraulic conductivity profiles in Figure 15 are related to instabilities in pH (outlined later, in section 4.6.2). In both cases, with and without the air entry option, the hydraulic conductivity reduces to below the saturated hydraulic conductivity. The reduction in hydraulic conductivity when the air entry option is turned on is due to the “hydraulic conductivity reduction due to chemistry” function as the soil chemistry conditions are not the optimum conditions so the hydraulic conductivity is reduced to below the saturated hydraulic conductivity. However the reductions in
hydraulic conductivity when the air entry option is turned off occur regardless of whether the hydraulic conductivity reduction due to chemistry option is turned on or not. The hydraulic conductivity profiles with the air entry value option and the “hydraulic conductivity reduction due to chemistry” option both turned off (Figure 4-17) look the same as results where with the air entry value option and the “hydraulic conductivity reduction due to chemistry” option both turned on (Figure 4-16). When the air entry option is turned on but the “hydraulic conductivity reduction due to chemistry” option is turned off, the hydraulic conductivity does not decrease below the saturated hydraulic conductivity once the profile is saturated (Figure 4-18).

Figure 4-15. Hydraulic conductivity profiles without the air entry option.
Figure 4-16. Hydraulic conductivity profiles with the air entry option.

Figure 4-17. Hydraulic conductivity profiles without the air entry option when the hydraulic conductivity reduction due to chemistry option is turned off.
Figure 4-18. Hydraulic conductivity profiles with the air entry option when the hydraulic conductivity reduction due to chemistry option is turned off.

Although there is an obvious improvement in the smoothness of the hydraulic conductivity curves and an improvement in response to the “hydraulic conductivity reduction due to chemistry” function, other model outputs did not suggest that there were instabilities in the model. For the cases presented here, with and without the air entry option, the pH profiles were the same, the water mass balance errors were approximately equal, the number of iterations for each time step were similar and the Peclet and Courant numbers for each time step were similar.

When the air entry option is turned off, the decreases in hydraulic conductivity occur when the pressure head is positive (Figure 4-19). The reason that this occurs is not known.
When the air entry option is used, the hydraulic conductivity decreases after saturation, even when the hydraulic conductivity reduction due to chemistry option is turned off. As this should not occur, it was decided that the air entry value should be applied for all simulations with fine textured soils. Choosing to use the air-entry option, will mean that while the soil profile is unsaturated, the hydraulic conductivity will be greater for a given pressure head when compared with the original hydraulic property function without the air-entry option (Figure 4-14).

### 4.6.2 Hydraulic Conductivity Reduction

The reductions in hydraulic conductivity that occur when the “hydraulic conductivity reduction due to chemistry” option is switched on are the result of a scaling parameter “r” which is in turn controlled by salt concentration, SAR and pH:

\[
K(h) = rK_sK_r = rK_sS_e^{1/2}[1-(1-S_e)^{1/m}]^2
\]  \hspace{1cm} (4.21)

\[
r(pH, \text{SAR, } C_0) = r_1(\text{SAR, } C_0)r_2(pH)
\]  \hspace{1cm} (4.22)
where $r_1$ represents the effect of ESP (SAR) and soil solution salinity, $C_0$, on the hydraulic conductivity and $r_2$ accounts for the effect of the soil solution pH on hydraulic conductivity.

The functions for the effects of SAR and salt concentration are based on the clay swelling model of McNeal (1968). Under this model, there is no reduction in hydraulic conductivity when the salt concentration is above 300 meq/L, below this value, the reductions in hydraulic conductivity depends on the soil ESP and the salt concentration (Figure 4-20). The function that relates soil sodicity and hydraulic conductivity includes a number of soil specific constants, the published values for these constants are used in the model (Simunek and Suarez 1997).

The pH of the soil solution reduces the hydraulic conductivity once the pH is greater than 6.83. The impact of pH on hydraulic conductivity is very significant, with a tenfold decrease in hydraulic conductivity when the pH is increased from 6.8 to 9.3 (Figure 4-21). It is important to note that the large measured effects of pH on hydraulic conductivity

![Figure 4-20. Effect of salt concentration and ESP on hydraulic conductivity.](image)
occurred at high SARs of 20 and 40 (Suarez, Rhoades et al. 1984). A maximum hydraulic conductivity of 2.2 cm/day is used for illustrative purposes.

The influence of soil solution chemistry on soil hydraulic conductivity can be seen clearly in the simulations used to mimic the column experiments. When gypsum is applied, the soil ESP is reduced towards zero but the pH of the solution causes a reduction the hydraulic conductivity.

### 4.6.3 Alkalinity of the Applied Solution

The significance of the alkalinity of the applied solution was discovered when attempting to use a saturated gypsum solution as the applied solution. When a solution containing only calcium and sulphate ions was used, and the alkalinity is equal to 0, the pH becomes acidic and the pH profiles become unstable (Figure 4-22). The instabilities in the pH profiles lead to instabilities in the hydraulic conductivity profiles only when the pH is in the range between 6.83 and 9.3 (Figure 4-23) i.e. the instabilities in the pH profile at day 5. The instabilities in pH and hydraulic conductivity can be improved by slightly increasing...
the alkalinity of the applied solution (Figure 4-24 and Figure 4-25). In this case, 0.1 meq/L of Mg and 0.1 meq/L of alkalinity were added to the gypsum solution. Addition of alkalinity to the solution causes an increase in the pH with results in a reduction in hydraulic conductivity. The alkalinity of the soil can also be increased by adding calcite into the soil profile but this also increases the pH which results in a lower hydraulic conductivity.

Figure 4-22. Instabilities in pH profiles when the alkalinity of the applied solution is zero.

Figure 4-23. Instabilities in hydraulic conductivity profiles when the alkalinity of the applied solution is zero.
4.6.4 Discretization

Grid discretization tests were used to determine a suitable grid spacing to use. Increasing the number of nodes decreased water mass balance errors. The number of nodes tested were 41, 81, 121, 241, 361, 481 and 601. The results presented here are for the simulations with gypsum as the applied solution. Increasing the number of nodes decreased the water
mass balance errors. The decreases in water mass balance errors were quite significant up until 240 nodes, followed by smaller changes after this point (Figure 4-26).

In contrast, increasing the number of nodes did not have a straightforward effect on chemical mass balance errors. When the number of nodes was increased, the chemical mass balance errors either increased or decreased. When 481 nodes were used, the chemical mass balance errors were much higher than when 241 nodes were used (Figure 4-27). However, when 601 nodes were used, both the water mass balance errors and chemical mass balance errors were reduced to a minimum.

![Figure 4-26. Water mass balance errors.](image-url)
Using a coarser grid also decreased the oscillations in pH down the soil profile. For the model with only gypsum in the applied solution, the number of nodes was decreased from 241 to 121 and the result was a decrease in the oscillations in pH as seen in the change from Figure 4-22 to Figure 4-28. The average pH values for each time interval with 121 nodes (Figure 4-28) were similar to the average pH values with 241 nodes (Figure 4-22).
4.6.5 Solute Transport Parameters

Sensitivity tests were performed to determine the influence of bulk density, dispersivity and the molecular diffusion coefficient on hydraulic conductivity. The starting point was the parameters used in the Simunek and Suarez (1997) model.

Within the range of expected bulk densities for the column experiments (1.3 to 1.5 g/cm³) and the field cases (up to 2.0 g/cm³), there was very little influence on hydraulic conductivity (Figure 4-7). It should be noted that the “bulk density” in this case is a parameter in the solute transport equation and thus alters hydraulic conductivity indirectly through effects on soil chemistry rather than altering the hydraulic property functions directly.
Within the range of expected dispersivities for the column experiments, between 0 and 4 cm, there was some small influence on hydraulic conductivity. Figure 4-29 shows the minor changes in hydraulic conductivity down the profile at day 1 when the dispersivity is varied. By 100 days, the hydraulic conductivity remains mostly unchanged as the dispersivity is varied. The hydraulic conductivity at the bottom of the soil profile at 100 days remains constant when the dispersivity is altered (Table 4-9).
Figure 4-29. Hydraulic conductivity profiles as affected by changes in longitudinal dispersivity.

Table 4-9. Hydraulic conductivity near the start and at the end of the simulation with varied dispersivity.

<table>
<thead>
<tr>
<th>Dispersivity (cm)</th>
<th>Hydraulic conductivity (cm/day)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0 cm, 1 day</td>
</tr>
<tr>
<td>0.0</td>
<td>1.14E+00</td>
</tr>
<tr>
<td>0.1</td>
<td>1.14E+00</td>
</tr>
<tr>
<td>0.4</td>
<td>1.14E+00</td>
</tr>
<tr>
<td>1.0</td>
<td>1.14E+00</td>
</tr>
<tr>
<td>4.0</td>
<td>1.13E+00</td>
</tr>
</tbody>
</table>

Changing the molecular diffusion coefficient had a significant influence on hydraulic conductivity at the beginning of the simulation. Figure 4-30 shows the changes in hydraulic conductivity down the profile at 1 day when the molecular diffusion coefficient was varied. By 100 days, the hydraulic conductivity remains mostly unchanged, particularly at the bottom of the soil profile (Table 4-10).
Figure 4-30. Hydraulic conductivity profiles as affected by changes in the molecular diffusion coefficient.

Table 4-10. Hydraulic conductivity near the start and at the end of the simulation with varied molecular diffusion coefficient.

<table>
<thead>
<tr>
<th>Diffusion coefficient (cm²/day)</th>
<th>Hydraulic conductivity (cm/day)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>0 cm,1 day</td>
</tr>
<tr>
<td></td>
<td>0 cm, 100 days</td>
</tr>
<tr>
<td></td>
<td>-4 cm,1 day</td>
</tr>
<tr>
<td></td>
<td>-4 cm, 100 days</td>
</tr>
<tr>
<td>0</td>
<td>1.34E+00</td>
</tr>
<tr>
<td>0.3</td>
<td>1.29E+00</td>
</tr>
<tr>
<td>3</td>
<td>1.20E+00</td>
</tr>
<tr>
<td>30</td>
<td>1.14E+00</td>
</tr>
<tr>
<td>300</td>
<td>1.13E+00</td>
</tr>
</tbody>
</table>
4.6.6 Comparison with Laboratory Experiment Results

The main purpose for simulating the laboratory experiments was to determine whether the overall trends and key processes could be simulated. The most important trend that needed to be simulated in the model is the increase in hydraulic conductivity to a maximum value when gypsum was applied, corresponding with decrease in exchangeable sodium. The model outputs that were used for comparison with laboratory experiment results include hydraulic conductivity, exchangeable sodium concentration and the sodium adsorption ratio (SAR) and pH of the soil solution.

The model results where gypsum was applied were used for comparisons. The gypsum solution with added alkalinity (0.1 meq/L of Mg and 0.1 meq/L of alkalinity) was used as this modification removed the instabilities in the hydraulic conductivity profiles (section 4.6.3).

The hydraulic conductivity gradually increased towards a maximum value. At day 1, the profile was saturated and increases in hydraulic conductivity started to occur at the top of the profile, as the gypsum was applied to this end. The maximum hydraulic conductivity was reached after 5 days (Figure 4-31 and Figure 4-32). In the experiments, it took at least 24 hours for flow out of the column to commence and more than ten days for the maximum hydraulic conductivity to be reached.
Figure 4-31. Hydraulic conductivity profiles when the modified gypsum solution was applied.

Figure 4-32. Hydraulic conductivity at the bottom of the soil profile when the modified gypsum solution was applied.

When the hydraulic conductivity reduction due to chemistry option was used, the maximum hydraulic conductivity that was reached during reclamation depended on the exchangeable sodium content of the soil as well as the salt concentration and pH of the soil solution. Based on the McNeal (1968) model that is incorporated into the hydraulic conductivity reduction function, once the soil ESP is less than 10, the hydraulic conductivity will be at a maximum. For this reason, the maximum hydraulic conductivity
is reached when the soil ESP is below 10, this corresponds to 29 meq/kg of exchangeable sodium in Figure 4-33 i.e. at day 5.

![Figure 4-33. Exchangeable sodium profiles when the modified gypsum solution was applied.](image)

The reason that the final hydraulic conductivity was below the saturated hydraulic conductivity was because the pH was not in the optimum range of between 6.83 and 9.3 (Section 4.6.2). At a pH of 7.3, the hydraulic conductivity would be predicted to be reduced from 2.2 cm/day (the saturated hydraulic conductivity entered for the soil hydraulic properties) to approximately 1.83 cm/day due to the effects of pH alone. This matches the hydraulic conductivity after day 4 shown in Figure 4-31.

In the laboratory experiments, the pH, electrical conductivity and cation concentrations of the leachate solutions were measured so these measured values could be compared to the model outputs. In the laboratory results, the pH of the leachate solution was dominated by the pH of the applied solution i.e. constant at around 7.2. This differs from the model results where there was a gradual reduction in pH starting at approximately 8.45 and finishing at approximately 7.3 (Figure 4-34). The final pH was similar to the measured values.
In the laboratory experiments, the initial Electrical Conductivity (EC) of the leachate solution was quite high, up to 10 mS/cm. The EC then gradually reduced to around the EC of the applied gypsum solution, 2.1 mS/cm. In the model results, the EC was initially quite low, approximately 0.25 (Figure 4-35). The EC then increased to around 2.5 before decreasing again to finish at approximately 2.1.
The measured SARs of the leachate solutions displayed a similar trend to the ECs. The initial SARs were approximately 55 and they gradually decreased to a minimum of approximately 0. In the model results, the initial SAR was at around 17 due to equilibration with the initial exchangeable cation concentrations where the ESP was around 17 (Figure 4-36). The SAR then gradually decreased towards 0.

![Figure 4-36. SAR profiles when the modified gypsum solution was applied.](image)

It is possible that the high initial measured EC and SAR in the leachate solutions were a result of soluble salts contained in the soil at the start of the experiment. As the presence of soluble salts was not included in the initial conditions for the modelling, the initial salt concentration for the soil solution was quite low due to the low salt concentration of the initial soil solution specified, “Burdekin River water”.

With the exception of the high initial concentrations of cations measured in the laboratory experiments, the trends in cation concentrations could be simulated (Figure 4-37). While the final magnesium and calcium concentrations in these simulations were very close to those measured in the laboratory experiments, the final sodium concentrations simulated were significantly higher than those measured in the laboratory experiments.
Chapter 4 Application of HYDRUS-1D to modelling sodic soil amelioration

Figure 4-37. Soil solution cation concentrations when the modified gypsum solution was applied.

4.7 Discussion

The HYDRUS-1D model with the UNSATCHEM module was used to simulate the amelioration of sodic soils. The key processes involved in sodic soil amelioration that were simulated in HYDRUS-1D were cation exchange and changes in hydraulic conductivity in response to changes in soil chemistry. The model outputs that were most suitable for showing the impacts of sodic soil amelioration were changes in exchangeable sodium concentrations, or sodium adsorption ratios, and changes in hydraulic conductivity.

The key functions in the UNSATCHEM model that were used for simulating amelioration of sodic soils were the hydraulic conductivity reduction function and the Gapon cation exchange functions. Using the “hydraulic conductivity reduction due to chemistry” option simulated the reduction in hydraulic conductivity that is believed to occur as a result of soil swelling in sodic soils. When this option was not used, the same chemical reactions occurred, including cation exchange which can cause changes in soil sodicity, but these changes have no impact on soil hydraulic properties.
HYDRUS-1D models were set up to replicate the published sodic soil amelioration models of Simunek and Suarez (1997). While some of the published results could be reproduced almost exactly, others could not be reproduced. The results that could not be reproduced were those where acidic waters were applied. As reclamation strategies with acidic water will not be included in models of the laboratory experiments and field cases, it was not critical to reproduce these examples. Of greater importance for further modelling was the reproduction of the examples where gypsum was used for reclamation. These examples were able to be reproduced almost exactly, with only small variations in timing. It is likely that the differences between the published results and the HYDRUS-1D results presented in this Chapter are the result of changes in the model code.

The published results represent hypothetical cases, produced to demonstrate the application of the hydraulic conductivity reduction function and to show the differences in reclamation time when different reclamation options were used. Therefore being able to reproduce the published results exactly was not necessary to validate that the HYDRUS-1D model is a suitable choice for modelling amelioration of sodic soils. The process of setting up models to replicate the published results was useful for obtaining a starting point for sensitivity testing and simulations of laboratory experiments and field cases.

When the simulations of the column experiments were set up, a number of issues were encountered. The first issue was instabilities in the model once the soil type was changed to “clay”. To improve the model stability, a modified form of the van Genuchten (1980) function, “the air entry option”, needed to be applied. This option was recommended in the HYDRUS-1D manual for fine textured soils. It was determined that using this option improves model stability in cases where other inputs, for example, the chemistry of the applied solution, had caused the model to become unstable.

When the “air entry option” was not used, it seemed as though there were problems with the “hydraulic conductivity reduction due to chemistry” option as the same reductions in hydraulic conductivity reductions were observed whether the “hydraulic conductivity reduction due to chemistry” option was turned on or not. When the “air entry option” was turned on, the expected behaviour for the “hydraulic conductivity reduction due to chemistry” option was observed, including no reductions in hydraulic conductivity when
the “hydraulic conductivity reduction due to chemistry” option was turned off. Because model instabilities are improved by using the “air entry option”, subsequent simulations for fine textured soils will be run using this option.

The second issue that was observed when simulations of the column experiments were set up was oscillations in the pH profiles. These oscillations lead to oscillations in the hydraulic conductivity profiles when the pH is in the range from 6.83 and 9.3. The oscillations in pH were observed only for certain compositions of applied solutions. It was determined that when no “alkalinity” was included in the composition of the applied solution, these oscillations started to occur. It was also determined that when no “alkalinity” was added, the pH became acidic. In the Simunek and Suarez (1997) example, negative alkalinitities were used to represent highly acidic solutions. The pH in UNSATCHEM is calculated using the alkalinity equation and a charge balance expression.

The third issue that was observed when comparisons were made with laboratory results was the strong dependence of hydraulic conductivity on the soil solution pH. The dependence of soil hydraulic conductivity on pH is controlled by the presence of variable charge minerals such as kaolinite and iron oxides (Sumner 1993). Increasing the pH can decrease the strength of the bonding between clay particles and lead to clay dispersion (Suarez, Rhoades et al. 1984). The degree of dependence of hydraulic conductivity on pH for the soils used in the laboratory experiments is not clear.

The strong dependence of hydraulic conductivity on pH in the model means that a small increase in the model calculated pH, for example, from 8 to 9, can cause a decrease in hydraulic conductivity of approximately sixty percent. This would correspond to large decreases in deep drainage fluxes.

Comparisons between the measured data from the column experiments and model outputs for the simulations of the column experiments where gypsum was used as the applied solutions indicated that the key processes were being captured. The key trend that was observable in the model outputs was the increase in soil hydraulic conductivity associated with the gradual decreases in soil sodicity, resulting from cation exchange. In the model, the time needed to reduce the soil sodicity to zero depended on the flow rates through the
soil and the cation exchange coefficients. These exchange coefficients were chosen based on the USDA (1954) equation and may not be accurate for the soils used in the laboratory experiments.

One of the trends that was noticed in the laboratory experiments but was not shown in the model results was the high initial EC and SAR of the soil solution. It is possible that this occurs due to the presence of soluble salts initially present in the soil. This aspect could be included in the model by increasing the salinity of the initial soil water or by adding precipitated minerals to the soil at the beginning of the simulation.

The grid independence tests lead to some surprising results. While it was expected that increasing the number of nodes would decrease mass balance errors, it was found that this was the case for water mass balance errors but not for chemical mass balance errors. The implications of this are that refining the mesh may not improve the model results and that care needs to be taken when determining a suitable discretization to use.

The results of the Simunek and Suarez (1997) replication models were found to be sensitive to a number of solute transport parameters, including bulk density, dispersivity and molecular diffusion coefficient. However, when sensitivity tests were conducted for the simulations of the column experiments, these solute transport parameters were determined to have less influence on the model outputs, possibly because of the much lower hydraulic conductivities. There were some small changes in the initial hydraulic conductivity when the dispersivity or molecular diffusion coefficient was altered but by the end of the simulations, the results were the same. Varying the “bulk density” in the solute transport parameters had very little impact on the hydraulic conductivity.

The model limitations need to be kept in mind for further modelling, particularly when using the simulations to make predictions about the long term impacts of sodic soil amelioration. Some of the key limitations that have been outlined in this Chapter and would need to be considered are the influence of the hydraulic conductivity reduction function, pH instabilities associated with the absence of alkalinity, the response of the simulations when the discretization is modified and the model instabilities that can occur with fine textured soils.
The hydraulic conductivity reduction function in HYDRUS-1D is a generic function yet the response of soil hydraulic conductivity to changes in soil chemistry would be expected to vary between different soils. The differences between the responses predicted by the model and the responses for specific soils need to be kept in mind. In particular, it would be important to take into account the influence of pH when studying the model predicted responses of sodic soils to different amelioration strategies.

Alkalinity needs to be included in the composition of applied solutions. Only small concentrations are needed to prevent oscillations in pH. For modelling the field cases, this will not be an issue as the irrigation water compositions will include alkalinity.

As there are limitations associated with increasing chemical mass balance errors with increasing numbers of nodes and model instabilities with fine textured soils, it will be necessary to adopt a number of measures. Any changes to the discretization of the model domain will require consideration of the impacts on chemical mass balance errors. It is has been shown that it is possible to cause an increase in chemical mass balance errors by increasing the number of nodes.

As there were model instabilities when fine textured soils were modelled, the “air-entry” option will be used for future simulations with fine textured soils. The use of this option will alter the hydraulic conductivity while the soil is unsaturated.

### 4.8 Conclusions

This Chapter has outlined the relevant functions in HYDRUS-1D that need to be applied for modelling sodic soil amelioration. These functions include the hydraulic property models, hydraulic conductivity reduction function and the cation exchange equations.

A series of models was set up to reproduce the published results of Simunek and Suarez (1997). While some of the published results could be reproduced using HYDRUS-1D, other results, particularly those with acidic solutions applied, could not be reproduced. It is
likely that the differences between the published results and the HYDRUS-1D results are
related to changes in the model code. The published results using gypsum for amelioration
could be reproduced reasonably well, with only small differences in timing.

Using simulations of the column experiments it was determined that the overall trends and
key chemical reactions could be adequately simulated. For the simulations where gypsum
was applied, the model results showed the gradual increase in hydraulic conductivity
associated with a decrease in soil sodicity that was also observed in laboratory
experiments.

Therefore, as the key processes could be simulated, HYDRUS-1D has been determined to
be applicable for modelling sodic soil amelioration. The scale and complexity of the
simulations need to be increased in order to make predictions about the potential impacts
of sodic soil amelioration on deep drainage. For future simulations, the model limitations
discussed in this Chapter will still be applicable. Key limitations highlighted in this
Chapter include the generalised hydraulic conductivity function, pH instabilities associated
with the absence of alkalinity, the response of the models when the discretization is
modified and the model instabilities that can occur with fine textured soils.
CHAPTER 5 – ASSESSING THE POTENTIAL IMPACTS OF SODIC SOIL AMELIORATION ON DEEP DRAINAGE

5.1 Introduction

In Chapter 4, the application of HYDRUS-1D for simulating the amelioration of sodic soils was tested and the short column results of Chapter 3 were modelled. The column experiment models in Chapter 4 are extended in the present Chapter to create “long columns” which represent ten metre deep uniform soil profiles. This is an appropriate scale for modelling the amelioration of sodic soils in the Lower Burdekin, where the depth of the upper clay layer can exceed 10 metres and high sodicity levels have been measured at depths of up to 2.5 metres, as described in Chapter 2. The potential impacts of sodic soil amelioration on groundwater recharge are assessed by examining changes in “deep drainage”, i.e. the drainage of water from the bottom of these “long columns”, corresponding to different soil properties, upper boundary conditions and amelioration treatments.

In Chapter 4, the initial soil hydraulic conductivity and exchangeable cation concentrations were set up to represent the soil used in the laboratory experiments. In the present Chapter, these properties are varied to assess the potential impacts of sodic soil amelioration under different soil conditions. The approach used to vary the soil hydraulic properties was to apply the Rosetta model as this enabled the influence of bulk density and soil texture on hydraulic conductivity and deep drainage to be studied.

In addition to investigating the impacts of soil properties, this Chapter includes comparisons between the two amelioration strategies commonly adopted in the Lower Burdekin, namely gypsum applications and “shandying” i.e. increasing the salinity of the irrigation water by mixing saline groundwater with river water. As gypsum is the most
widely used strategy, the influence of gypsum applications on deep drainage is studied for a range of soil conditions. These simulations use a constant head upper boundary condition to determine the maximum depth of changes in soil sodicity and hydraulic conductivity that might be expected to occur in response to continuous treatments over a 10 year period.

As continuous applications of gypsum represent an extreme situation, further simulations were carried out using an atmospheric upper boundary condition to investigate the combined influence of sporadic gypsum applications, irrigation applications, seasonal rainfall and evaporation. Using these simulations, the impacts of different gypsum application frequencies was also investigated.

To determine the importance of considering surface geometry features, in particular irrigation furrows, when simulating sodic soil amelioration, the HYDRUS-1D models with a seasonal upper boundary condition were extended to 2D. These simulations were set up using HYDRUS-2D/3D (Simunek, van Genuchten et al. 2006). Using these 2D models, the influence of localised gypsum applications within irrigation furrows was investigated.

Lastly, the limitations of applying these models and interpretation of simulations are discussed.

### 5.2 Soil Hydraulic Properties

The van Genuchten (1980) model was used for all of the simulations in this Chapter. Two methods were applied to select hydraulic properties. The first method was to select the “clay” soil from the HYDRUS catalogue and reduce the saturated hydraulic conductivity from 4.8 to 2.2 cm/day. This method was recommended by Suarez (2001) for the simulation of sodic soil amelioration and was used in Chapter 4 for simulation of the column experiments. The second method that was applied in this Chapter was to use the Rosetta model. This method was used to enable the impacts of soil texture and bulk density on deep drainage to be tested.
The parameters included in the HYDRUS catalogue are those from Carsel and Parrish (1988) and represent approximate averages for the different textural classes. The soil database compiled by Carsel et al. (1988) was used to compute these hydraulic parameters. This soil database includes the measurements of properties of almost 3000 soils from the USA.

The Rosetta computer program (Schaap, Leij et al. 2001) implements pedotransfer functions to predict hydraulic parameters from soil textural distribution, bulk density and two water retention points. Using this method, hydraulic parameters can also be predicted using less information, for example, soil texture and bulk density only. The calibration data used for the Rosetta database contained 2134 measurements for water retention and 1306 measurements for $K_s$ (Schaap and Leij 1998). The data was obtained from a large number of sources and involved agricultural and non-agricultural soils in temperate climate zones of the northern hemisphere.

As both methods are based on data mostly from the USA, differences between these soils and Australian soils may result in inaccuracies in using the predicted parameters to represent Australian soils. In addition, the majority of the measured hydraulic properties for both of these methods were for soils with less than 60% clay. The predictions may therefore be expected to be less applicable for soils with higher clay contents. However, these methods provide tools for estimating hydraulic parameters in cases where measured hydraulic property data are not available.

5.3 HYDRUS-1D Soil Profile Conceptualisation

The 1D soil profile simulations described in this Chapter simulate a vertical soil profile depth of ten metres over a 10 year period to assess the impacts of long term sodic soil amelioration on deep drainage for different soil conditions and amelioration treatments. Initially uniform soil properties down the profile, including soil chemistry and soil hydraulic properties, were assumed. The hydraulic conductivity reduction function
(Simunek, van Genuchten et al. 2005) (described in section 4.6.2) was used for all of the simulations in this Chapter.

The discretization used in the simulations in this Chapter were as follows. Evenly spaced grid cells of 1 cm height were used for the ten metre profile. This resulted in the maximum number of grid cells possible in HYDRUS-1D version 4.09, i.e. 1000 grid cells. Reducing the initial and minimum time steps reduces mass balance errors and improves model stability. Therefore, the smallest time steps chosen in Chapter 4 were also used in this Chapter. The initial time step and the minimum time step were set at $1 \times 10^{-6}$ days and $1 \times 10^{-7}$ days respectively. The maximum time step was set at 1 day.

The initial conditions for the these simulations were the same as those used for the column simulations in Chapter 4, with the exception of the solute transport parameter “bulk density” which was increased to 2.0 g/cm$^3$ to more closely represent the bulk density of the field soils. However, as the variations in hydraulic conductivity were found to be insensitive to the solute transport parameter “bulk density” in Chapter 4, (see section 4.6.5), this parameter was held constant for the simulations in this Chapter. As before, the van Genuchten (1980) model of soil hydraulic properties was used with the air entry option (Simunek, van Genuchten et al. 2005) (see section 4.2.1). For the lower boundary condition, a free drainage condition was applied.

### 5.3.1 Constant Head Upper Boundary Condition Simulations

Using a constant head upper boundary condition to simulate continuous application of a saturated gypsum solution, modelling experiments were performed to investigate the influence of the bulk density, soil texture and initial soil sodicity on the extent, depth and timing of soil reclamation and corresponding changes in deep drainage rates. The variations in soil properties represent the range that might be expected within the BHWSS. The impacts of bulk density and soil texture were determined by using the Rosetta model to estimate soil hydraulic properties in the simulations.

Continuous application of a saturated gypsum solution was used to investigate the maximum impacts on potential deep drainage over a ten year period. The impacts of the
saturated gypsum solution were then compared with the impacts of varying irrigation water quality i.e. river water vs. “shandied” irrigation water. The degree of reductions in soil sodicity and increases in hydraulic conductivity were used as measures of reclamation.

5.3.1.1 The Effect of Bulk Density on Deep Drainage

To determine the influence of bulk density on soil hydraulic properties and deep drainage, the Rosetta model was used to estimate the soil hydraulic properties by holding the soil texture constant and varying the soil bulk density between 1.3 and 2.0 g/cm$^3$. Bulk densities that have been measured in the Lower Burdekin range from values as low as 1.32 g/cm$^3$ (Eldridge 2003) to approximately 1.9 g/cm$^3$ for the soils sampled for use in laboratory experiments (see section 3.3). The soil texture was based on the average particle size distribution of the sodic clays used in the laboratory experiments: sand = 18%, silt = 12% and clay = 70%. The changes in hydraulic conductivity when bulk density was varied are shown in Figure 5-1 and Figure 5-2. When the bulk density was increased, the hydraulic conductivity decreased as would be expected due to the decrease in pore space (Assouline 2006). The hydraulic properties when the bulk density was varied are presented in Table 5-1, the parameters used for the column models in Chapter 4 are included for comparative purposes.
Chapter 5  
Assessing the potential impacts of sodic soil amelioration on deep drainage

Figure 5-1. Changes in hydraulic conductivity curves as a function of soil bulk density.

Figure 5-2. Changes in saturated hydraulic conductivity as a function of soil bulk density.
Table 5-1. Soil hydraulic properties for a clay soil, used in the column experiment simulations.

<table>
<thead>
<tr>
<th>Method</th>
<th>Details</th>
<th>( \theta_r )</th>
<th>( \theta_s )</th>
<th>( \alpha ) (cm(^{-1}))</th>
<th>n</th>
<th>( K_s ) (cm/day)</th>
<th>1</th>
</tr>
</thead>
<tbody>
<tr>
<td>HYDRUS catalogue soil type</td>
<td>“Clay” with a modified saturated hydraulic conductivity</td>
<td>0.068</td>
<td>0.38</td>
<td>0.008</td>
<td>1.09</td>
<td>2.2</td>
<td>0.5</td>
</tr>
<tr>
<td>Rosetta model, bulk density varied</td>
<td>BD = 1.3g/cm(^3)</td>
<td>0.106</td>
<td>0.51</td>
<td>0.021</td>
<td>1.23</td>
<td>16.0</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>BD = 1.4g/cm(^3)</td>
<td>0.103</td>
<td>0.48</td>
<td>0.020</td>
<td>1.21</td>
<td>11.5</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>BD = 1.5g/cm(^3)</td>
<td>0.099</td>
<td>0.45</td>
<td>0.020</td>
<td>1.19</td>
<td>8.2</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>BD = 1.6g/cm(^3)</td>
<td>0.096</td>
<td>0.42</td>
<td>0.020</td>
<td>1.18</td>
<td>5.8</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>BD = 1.7g/cm(^3)</td>
<td>0.094</td>
<td>0.39</td>
<td>0.021</td>
<td>1.17</td>
<td>4.0</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>BD = 1.8g/cm(^3)</td>
<td>0.091</td>
<td>0.36</td>
<td>0.022</td>
<td>1.16</td>
<td>2.7</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>BD = 1.9g/cm(^3)</td>
<td>0.090</td>
<td>0.34</td>
<td>0.022</td>
<td>1.16</td>
<td>1.8</td>
<td>0.5</td>
</tr>
<tr>
<td></td>
<td>BD = 2.0/cm(^3)</td>
<td>0.088</td>
<td>0.32</td>
<td>0.023</td>
<td>1.16</td>
<td>1.2</td>
<td>0.5</td>
</tr>
</tbody>
</table>

5.3.1.2 The Effect of Clay Content on Deep Drainage

The proportion of clay in the sodic soils of the Lower Burdekin is variable. For the most common soil types in the BHWSS, namely cracking clays and duplex soils, the soil texture typically ranges from a sandy loam to a heavy clay, with light and medium clays also common (Donnollan 1991). To determine the influence of variations in clay percentage on soil hydraulic properties and deep drainage using the Rosetta model, the bulk density was held constant at 1.8 g/cm\(^3\) and the soil texture was varied. To simplify the variations in soil texture, the silt % was kept constant at 10% so only the clay % and sand % needed to be adjusted while keeping the total at 100%. The range of clay percentages investigated was from 10% to 90%. This range covered soil types from a loamy sand to a clay soil (Figure 5-3 and Table 5-2).
Figure 5-3. USDA soil texture diagram showing the range of clay soil types analysed in this study (dotted line).

Table 5-2. Varied clay percentages used with corresponding textures.

<table>
<thead>
<tr>
<th>% silt</th>
<th>% sand</th>
<th>% clay</th>
<th>Soil texture based on USDA soil texture triangle</th>
</tr>
</thead>
<tbody>
<tr>
<td>10</td>
<td>80</td>
<td>10</td>
<td>Loamy sand / sandy loam</td>
</tr>
<tr>
<td>10</td>
<td>70</td>
<td>20</td>
<td>Sandy loam / sandy clay loam</td>
</tr>
<tr>
<td>10</td>
<td>60</td>
<td>30</td>
<td>Sandy clay loam</td>
</tr>
<tr>
<td>10</td>
<td>50</td>
<td>40</td>
<td>Sandy clay</td>
</tr>
<tr>
<td>10</td>
<td>40</td>
<td>50</td>
<td>Clay</td>
</tr>
<tr>
<td>10</td>
<td>30</td>
<td>60</td>
<td>Clay</td>
</tr>
<tr>
<td>10</td>
<td>20</td>
<td>70</td>
<td>Clay</td>
</tr>
<tr>
<td>10</td>
<td>10</td>
<td>80</td>
<td>Clay</td>
</tr>
<tr>
<td>10</td>
<td>0</td>
<td>90</td>
<td>Clay</td>
</tr>
</tbody>
</table>
The hydraulic properties for the soils with 10% or 20% clay, namely the loamy sand / sandy loam and the sandy loam / sandy clay loam (Table 5-2), were significantly different from the hydraulic properties for the soil types with clay contents of 30% or higher (Figure 5-4). Specifically, the van Genuchten (1980) parameter “n” and the saturated hydraulic conductivity were much larger for soils with clay contents below 30% (Table 5-3 and Figure 5-5). For the soils with clay contents above 30%, the variations in hydraulic properties were minimal (Figure 5-4 and Table 5-3).

Table 5-3. Hydraulic properties when clay percentage is varied.

<table>
<thead>
<tr>
<th>Method</th>
<th>Details</th>
<th>( \theta_r )</th>
<th>( \theta_s )</th>
<th>( \alpha ) (cm(^{-1}))</th>
<th>n</th>
<th>( K_s ) (cm/day)</th>
<th>l</th>
</tr>
</thead>
<tbody>
<tr>
<td>HYDRUS catalogue soil type</td>
<td>“Clay” with a modified saturated hydraulic conductivity</td>
<td>0.068</td>
<td>0.38</td>
<td>0.008</td>
<td>1.09</td>
<td>2.2</td>
<td>0.5</td>
</tr>
<tr>
<td>10%</td>
<td></td>
<td>0.044</td>
<td>0.308</td>
<td>0.0409</td>
<td>1.526</td>
<td>29.1</td>
<td>0.5</td>
</tr>
<tr>
<td>20%</td>
<td></td>
<td>0.050</td>
<td>0.318</td>
<td>0.0346</td>
<td>1.250</td>
<td>8.1</td>
<td>0.5</td>
</tr>
<tr>
<td>30%</td>
<td></td>
<td>0.058</td>
<td>0.330</td>
<td>0.0317</td>
<td>1.171</td>
<td>4.2</td>
<td>0.5</td>
</tr>
<tr>
<td>40%</td>
<td></td>
<td>0.069</td>
<td>0.342</td>
<td>0.0313</td>
<td>1.146</td>
<td>3.4</td>
<td>0.5</td>
</tr>
<tr>
<td>50%</td>
<td></td>
<td>0.080</td>
<td>0.353</td>
<td>0.0298</td>
<td>1.143</td>
<td>3.3</td>
<td>0.5</td>
</tr>
<tr>
<td>60%</td>
<td></td>
<td>0.088</td>
<td>0.359</td>
<td>0.0262</td>
<td>1.150</td>
<td>3.2</td>
<td>0.5</td>
</tr>
<tr>
<td>70%</td>
<td></td>
<td>0.092</td>
<td>0.364</td>
<td>0.0222</td>
<td>1.159</td>
<td>2.9</td>
<td>0.5</td>
</tr>
<tr>
<td>80%</td>
<td></td>
<td>0.096</td>
<td>0.366</td>
<td>0.0184</td>
<td>1.167</td>
<td>2.5</td>
<td>0.5</td>
</tr>
<tr>
<td>90%</td>
<td></td>
<td>0.100</td>
<td>0.368</td>
<td>0.0155</td>
<td>1.193</td>
<td>1.9</td>
<td>0.5</td>
</tr>
</tbody>
</table>
Figure 5-4. Hydraulic conductivity curves as a function of clay percentage.

Figure 5-5. Saturated hydraulic conductivity as a function of clay percentage.
5.3.1.3 The Effect of Soil Sodicity on Deep Drainage

As soil sodicity is the aspect of soil chemistry that is expected to have the most influence on soil hydraulic properties and deep drainage rates, the influence of initial exchangeable sodium percentage was studied using the “clay” HYDRUS catalogue hydraulic properties with a saturated hydraulic conductivity of 2.2 cm/day (Table 5-3). The range of initial Exchangeable Sodium Percentages (ESPs) tested included the maximum and minimum ESP values measured in the Mulgrave area of the BHWSS, based on measurements from 44 sites. This data was accessed from the Queensland Department of Environment and Resource Management’s SALI database on 28/02/2008. The ESPs tested also included two-fold and fourfold increases and decreases of the site 1 ESP of 16.9. The ESP was varied by maintaining the CEC and adjusting the exchangeable cation concentrations.

For the other simulations described in this Chapter, the exchangeable cation concentrations were kept the same as those for the “column simulations” in Chapter 4. These concentrations were 102 meq/kg for Ca, 136 meq/kg for Mg, 49 meq/kg for Na and 3 meq/kg for K. This is the “original base case”. For this series of simulations, where the ESP was varied, the exchangeable cation concentrations had to be varied. The approach used was to make the calcium and magnesium concentrations equal to each other and the potassium concentrations equal to zero, so that the CEC could be maintained and the cation concentrations could be adjusted easily for each ESP. For example, to maintain the same ESP as the base case of 16.9, the exchangeable cation concentrations were adjusted as follows: the Ca concentration was increased to 120.5 meq/kg, the Mg concentration was decreased to 120.5 meq/kg, the K concentration was decreased to 0 and the concentration of Na was maintained at 49 meq/kg (Table 5-4). This is the “modified base case”.

Potassium concentration has previously been found to have minimal impact on soil physical properties (USDA 1954) and because potassium only constitutes a small proportion of the exchange complex for the soils of interest, it is not considered here. To determine the influence of these adjustments of exchangeable cation concentrations on the deep drainage results, the “original base case” was compared with the “modified base case” where the ESP and CEC were kept the same but the concentrations of exchangeable calcium and magnesium varied, and the concentration exchangeable potassium was
reduced to zero. For these two cases, the deep drainage rates were found to be almost identical.

Table 5-4. Exchangeable cation concentration when the ESP is varied. An ESP of 16.9 represents the base case.

<table>
<thead>
<tr>
<th>ESP</th>
<th>Exchangeable Cation Concentrations (meq/kg)</th>
<th>CEC (meq/kg)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Na</td>
<td>Ca</td>
</tr>
<tr>
<td>67.6</td>
<td>196.0</td>
<td>47.0</td>
</tr>
<tr>
<td>47.8</td>
<td>138.6</td>
<td>75.7</td>
</tr>
<tr>
<td>33.8</td>
<td>98.0</td>
<td>96.0</td>
</tr>
<tr>
<td><strong>16.9</strong></td>
<td><strong>49.0</strong></td>
<td><strong>120.5</strong></td>
</tr>
<tr>
<td>8.5</td>
<td>24.7</td>
<td>132.7</td>
</tr>
<tr>
<td>4.2</td>
<td>12.2</td>
<td>138.9</td>
</tr>
<tr>
<td>0.7</td>
<td>2.0</td>
<td>144.0</td>
</tr>
</tbody>
</table>

### 5.3.1.4 The Effect of Amelioration Treatments on Deep Drainage

A number of amelioration treatments were simulated using the “clay” HYDRUS catalogue hydraulic properties with a saturated hydraulic conductivity of 2.2 cm/day (Table 5-3). The amelioration treatments simulated represent the end members for irrigation water quality that might be applied in the field in addition to a saturated gypsum solution. The first irrigation water composition, “river water” is based on the average composition of Burdekin River water at Clare, as described in Chapter 3. The second option, “gypsum dissolved in river water” consists of the same river water, to which gypsum has been added in excess to achieve a solution saturated with gypsum. The third option, “shandied water” consists of a “shandy” where 20% of the local Mulgrave saline groundwater is mixed with 80% of the Burdekin River water. The approximate electrical conductivities (ECs) of these solutions are 200 µS/cm for the river water and 2000 µS/cm for the gypsum dissolved in river water and the shandied water. The compositions of these three solutions are provided in Table 5-5. Constant head conditions were used so each of these solutions was effectively applied continuously.
Table 5-5. Composition of applied solutions, in meq/L.

<table>
<thead>
<tr>
<th></th>
<th>Ca</th>
<th>Mg</th>
<th>Na</th>
<th>K</th>
<th>Alkalinity</th>
<th>SO(_4)</th>
<th>Cl</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. “river water”</td>
<td>0.75</td>
<td>0.64</td>
<td>0.78</td>
<td>0.08</td>
<td>1.66</td>
<td>0.11</td>
<td>0.59</td>
</tr>
<tr>
<td>2. “gypsum dissolved in river water”</td>
<td>31.75</td>
<td>0.64</td>
<td>0.78</td>
<td>0.08</td>
<td>1.66</td>
<td>31.11</td>
<td>0.59</td>
</tr>
<tr>
<td>3. “shandied water”</td>
<td>2.56</td>
<td>3.90</td>
<td>14.72</td>
<td>0.08</td>
<td>3.70</td>
<td>0.56</td>
<td>17.00</td>
</tr>
</tbody>
</table>

5.3.2 Constant Head Upper Boundary Condition Results

The results presented in this section show:

i) the impacts of changes in soil hydraulic properties on potential deep drainage and

ii) the impacts of changes in initial soil sodicity and amelioration treatments on soil sodicity, hydraulic conductivity and deep drainage.

The final results are for the end of the 3650 day simulation period in each case.

5.3.2.1 The Effect of Bulk Density on Deep Drainage

As the bulk density of the soil was increased, the hydraulic conductivity decreased and the corresponding deep drainage rates decreased along with the time taken to reach the maximum deep drainage rate (Figure 5-6). As a result, the time taken for a response to the applied gypsum solution was increased as the bulk density was increased. The total deep drainage after 10 years was also reduced as the bulk density was increased (Figure 5-7). When the bulk density was increased between 1.4 g/cm\(^3\) and 2.0 g/cm\(^3\), there was a tenfold decrease in the total deep drainage.
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Figure 5-6. Deep drainage as a function of bulk density.

Figure 5-7. Total deep drainage as a function of bulk density.
5.3.2.2 The Effect of Clay Content on Deep Drainage

Increases in clay percentage resulted in similar trends in deep drainage to the trends with increases in bulk density. As the clay percentage is increased, the maximum deep drainage rate and the time taken to reach the maximum deep drainage rate both decreased. There is a large jump in the deep drainage rates when the clay percentage is decreased from 40\% to 20\% which corresponds to a change in soil type from a “loam” to a “clay”. The total deep drainage after 10 years was also reduced as the clay content was increased (Figure 5-9). There was a fourfold increase in deep drainage when the clay content was reduced from 20\% to 10\% and a further twofold increase in deep drainage when the clay content was reduced from 40\% to 20\%. The results for 30\% clay are not shown here as this simulation did not converge so results for the full ten year period are not available.

![Figure 5-8. Deep drainage as a function of clay percentage.](image-url)
5.3.2.3 The Effect of Soil Sodicity on Deep Drainage

Sodicity reductions are expected to lead to increases in hydraulic conductivity and deep drainage rates. In HYDRUS, the relationship between ESP and hydraulic conductivity is based on the equations of McNeal (1968). According to these equations, for a gypsum saturated solution with a salt concentration of 33 meq/L, the hydraulic conductivity is at its maximum once the ESP is reduced to below 20. Thus, according to these functions, further reductions in ESP would not cause further increases in hydraulic conductivity. These relationships are soil specific, so it is possible that further reductions in sodicity below 20 may lead to further increases in hydraulic conductivity. This is supported by the results in Chapter 3. For this reason, both ESP profiles and deep drainage rates are presented in this Chapter as measures of sodic soil reclamation.

The initial exchangeable sodium concentrations were varied and the influence on the final exchangeable sodium concentrations determined. When the initial ESP was set at low values of 0.7, 4.2 or 8.5, the ESPs were reduced to approximately 0.3 throughout the soil profile after 10 years of continuous leaching with a saturated gypsum solution (Figure 5-10). However, when the initial ESPs were set at 16.9 or above, most of the profile remained at concentrations close to the initial concentration (Figure 5-10). This means that
the “reclamation” was mostly limited to depths of less than 1 m, even after ten years of treatment.

![Figure 5-10. Final Exchangeable Sodium Percentage (ESP) as a function of initial ESP.](image)

Figure 5-10 shows that for initial ESPs equal to or less than 16.9, deep drainage rates increased considerably over the 10 year simulation period. When initial ESPs were greater than 16.9, the deep drainage rates remained close to zero. Most of the change in deep drainage rates in response to initial ESP takes place in the first five years (Figure 5-11). The total deep drainage over the ten year period is inversely related to the initial ESP (Figure 5-12).
Chapter 5 Assessing the potential impacts of sodic soil amelioration on deep drainage

Figure 5-11. Changes in deep drainage rates over time with varied initial exchangeable sodium concentrations.

Figure 5-12. Total deep drainage after ten years as a function of initial ESP.
For initial ESPs of less than 16.9, the final deep drainage rates were not sensitive to the initial ESP. By 5 years, the deep drainage rates for an initial ESP of 16.9 had reached a maximum (Figure 5-11). Between 5 and 10 years, the ESP was further reduced (Figure 5-13) but the hydraulic conductivity did not increase any further (Figure 5-14). These results reflect the functions of (McNeal 1968) where ESPs below 20 do not cause reductions in hydraulic conductivity for salt concentrations above 30 meq/L.

Figure 5-13. Exchangeable sodium concentrations over time for an initial ESP of 16.9.
Increases in hydraulic conductivity were expected with decreases in sodicity, increases in salt concentrations and decreases in pH. The greatest increase in deep drainage occurred with an initial ESP of 16.9, where the deep drainage rate increased from 0.2 cm/day after one year to close to 1.45 cm/day after five years (Figure 5-11). The final EC of the soil solution remained close to the EC of the applied solution for the initial ESPs equal to or less than 16.9. However, the final pHs for the initial ESPs of less than 16.9 were approximately 0.1 pH units greater than the final pH for the initial ESP of 16.9 (data not shown). Based on the hydraulic conductivity reduction function used in HYDRUS (as described in section 4.6.2), an increase in pH of 0.1 pH units would cause a decrease in hydraulic conductivity of up to 20% and this would be expected to cause a corresponding decrease in deep drainage rates for the lowest initial ESPs.
5.3.2.4 The Effect of Amelioration Treatments on Deep Drainage

Both the treatment with “gypsum dissolved in river water” and the treatment with “shandied water” resulted in reductions in the exchangeable sodium concentration in the soil profile. For the case with “gypsum dissolved in river water”, the exchangeable sodium concentrations were reduced to less than 1.2 meq/100g in the top 7.55 metres of the soil profile by the end of the 10 year simulation period. In the bottom part of the soil profile, from 8.58 metres to 10 metres, the exchangeable sodium concentrations remained above 48 meq/100g (Table 5-6), close to the initial concentration of 49 meq/100g.

For the case with “shandied water”, most of the reduction in sodicity occurred in the top 1 metre of the profile and the exchangeable sodium concentrations in this top 1 metre were between 31 and 34 meq/100g. The decrease was therefore much less for the “shandied water” when compared with the “gypsum dissolved in river water”. In the case where “river water” was applied, there was very little change in the exchangeable sodium concentrations over the 10 years simulated (Table 5-6) and the concentrations remained close to the initial concentration of 49 meq/100g. The only reductions in sodicity that occurred for this option occurred in the top 0.6 metres.

The final hydraulic conductivity was constant down the profile for all of the treatment options tested, with the exception of the “river water” option which resulted in a higher hydraulic conductivity of close to 1 cm/day, where the sodicity had been reduced. The final hydraulic conductivity was the highest when “gypsum dissolved in river water” was applied. The “shandied water” resulted in an approximately a five fold increase in hydraulic conductivity when compared with “river water” alone.

The final hydraulic conductivity is expected to be influenced by the ESP of the soil and the EC and pH of the soil solution. The ESPs in the top metre of the soil profile were reduced the most when the gypsum in river water was applied and reduced the least when shandied water was applied.

The EC of “gypsum dissolved in river water” was similar to the EC of the “shandied water” and the EC of both of these solutions was almost ten times greater than that of the
“river water”. The final pH for the “river water” and the “shandied water” was approximately 8.4 whereas the approximate pH of the “gypsum in river water” was 7.8. So the combination of reduced ESP and pH and increased EC with the “gypsum in river water” leads to the highest hydraulic conductivity.

Table 5-6. Sodicity (exchangeable sodium concentration), hydraulic conductivity and total deep drainage for the three different treatments.

<table>
<thead>
<tr>
<th></th>
<th>Final Exchangeable Sodium Concentration at 1 metre (meq/kg)</th>
<th>Final Exchangeable Sodium Concentration at 10 metres (meq/kg)</th>
<th>Final Hydraulic Conductivity at 1 metre (cm/day)</th>
<th>Final Hydraulic Conductivity at 10 metres (cm/day)</th>
<th>Total Deep Drainage (cm)</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. “river water”</td>
<td>48.76</td>
<td>48.76</td>
<td>0.22</td>
<td>0.22</td>
<td>771</td>
</tr>
<tr>
<td>2. “gypsum dissolved in river water”</td>
<td>1.04</td>
<td>48.57</td>
<td>1.47</td>
<td>1.45</td>
<td>3670</td>
</tr>
<tr>
<td>3. “shandied water”</td>
<td>33.74</td>
<td>48.74</td>
<td>0.96</td>
<td>0.92</td>
<td>2320</td>
</tr>
</tbody>
</table>

The deep drainage rates for all of the treatment options start at close to zero at the beginning of the simulations, when the soil profiles are dry. At between 150 and 180 days, the deep drainage rates increase to around 0.2 cm/day (Figure 5-15). This corresponds to the time period taken to fully saturate the soil profile. The treatment with “river water” took the longest amount of time to saturate and show an increase in deep drainage rate. At approximately 850 days, the deep drainage rates for the “shandied water” and “gypsum dissolved in river water” treatments showed a further increase in deep drainage rates. The final deep drainage rates correspond to the final hydraulic conductivities for each applied solution which are the result of changes in soil chemistry.
5.3.3 Seasonal Upper Boundary Condition Simulations

A series of simulations were set up using an “atmospheric upper boundary condition with surface layer” (Simunek, van Genuchten et al. 2005). The purpose of these simulations was to explore the influence of seasonal rainfall and different frequencies of gypsum application on soil properties and deep drainage rates. For these simulations, daily irrigation and rainfall data were combined as only one column of data is allowed for applied water. A separate column is used to specify solution chemistry so irrigation water chemistry was specified for the days with irrigation.

In HYDRUS, the atmospheric boundary condition is implemented using prescribed head or prescribed flux conditions. The boundary condition may change from prescribed head to prescribed flux conditions and vice-versa. The actual flux is dependent on the transient soil moisture conditions near the surface (Simunek, van Genuchten et al. 2005).
The surface layer is used to specify the maximum thickness of the surface water layer before surface runoff is initiated. For this study, the thickness of the surface water layer was reduced to a low value 0.1 cm to enable the model to run with minimal impact on deep drainage.

The simulations with an atmospheric upper boundary condition were typically unstable and often did not converge when they were first set up. When the hydraulic conductivity function was applied, hydraulic conductivities varied rapidly at different positions in the soil profile and this may make calculations of water fluxes more difficult during wetting or drying. In order to get the simulations to run, the hydraulic properties were changed to those for a “clay loam” provided by the HYDRUS library hydraulic properties. This would affect the magnitude of deep drainage predicted (by altering both the proportion of rainfall and irrigation that becomes runoff and the hydraulic conductivity of the soil) but still allow exploration of the relative effects of rainfall, gypsum application and evaporation on deep drainage.

### 5.3.3.1 Seasonal Inputs

The seasonal atmospheric inputs were based on one year of recorded irrigation rates at site 1 and rainfall measurements from the nearby “Clare” rainfall station (see section 2.2). Most of the irrigation occurs between July and December, before the wet season (Figure 5-16). This pattern of irrigation was repeated over a ten year period, between 1999 and 2009, with actual rainfall for each year included to make up the seasonal boundary condition. By including ten years of rainfall data, “wet” years and “dry” years could be simulated and the impacts of rainfall variations could be tested as the irrigation applications were kept constant. The rainfall data (Figure 5-17) was downloaded from the Bureau of Meteorology’s SILO website ([http://www.bom.gov.au/silo/](http://www.bom.gov.au/silo/)).
Figure 5-16. Daily irrigation applications at site 1 in 1999.

Figure 5-17. Daily rainfall at "Clare" from 1999 to 2009.
The composition of the irrigation water was kept the same as “river water”, as described in section 5.3.1.4. The rainfall composition used with these seasonal boundary condition simulations contained 10% of the salinity of the river water. The “gypsum dissolved in river water” which was described in section 5.3.1.4, was applied in October, during the middle of the irrigation season. For the years where gypsum was applied, it was applied with the three irrigation applications in October. Using this application method, the equivalent quantity of gypsum applied in October was approximately 7 tonnes / hectare, which is within the range of gypsum application rates normally recommended for sodic soils of 5 to 10 tonnes / hectare (Nelson, Ham et al. 2001). The gypsum application frequencies tested were: no gypsum, one application in ten years, two applications in ten years, three applications in ten years and ten applications in ten years i.e. yearly gypsum applications.

Evaporation data for the Clare site was also downloaded from the Bureau of Meteorology’s SILO website. The measured daily evaporation typically ranges from 4 mm to 10 mm (Figure 5-18). The maximum evaporation rates occur between November and February. Simulations were run with no evaporation and with 10% and 100% of measured evaporation rates to determine the influence of evaporation on deep drainage rates.

Figure 5-18. Daily evaporation at "Clare" from 1999 to 2009.
5.3.4 Seasonal Upper Boundary Condition Results

The results presented for simulations with a seasonal boundary condition demonstrate:

(i) The depth and timing of soil sodicity reductions resulting from different frequencies of gypsum applications, interspersed with irrigation and rainfall; and

(ii) The combined impacts of gypsum application frequency and seasonal variations in irrigation, rainfall and evaporation on potential deep drainage.

5.3.4.1 Soil Properties in the Root Zone

When gypsum dissolved in river water was applied at yearly intervals at a rate of approximately 7 tonnes per hectare per year, decreases in soil sodicity only occurred in the top 50 cm of the soil profile, even after 10 years of gypsum applications. When gypsum was applied once a year, there was a stepwise decrease in sodicity beginning with the top 10 cm of the profile and then gradually moving deeper in the profile with each gypsum application (Figure 5-19). In the case where gypsum was applied only once, there was a decrease in sodicity in the top 10 cm following the gypsum application (Figure 5-20). In subsequent years there were small decreases in sodicity in the top 10 cm.

Figure 5-19. Exchangeable sodium concentrations when gypsum was applied once a year.
When no gypsum was applied, there were still small decreases in exchangeable sodium concentration (Figure 5-21). These decreases in sodicity only occurred in the top few centimetres of the soil profile and must have been due to cation exchange with the rainfall and irrigation water followed by leaching of the sodium ions. The extent to which cation exchange occurs depends on the exchangeable cation concentrations of the soil, the cation concentrations in the applied solutions and the cation exchange coefficients of the soil.

When no gypsum is applied, changes in soil salinity are primarily due to flushing by rainfall or the addition of salts from irrigation water. The changes in salt concentration of
the soil solution in response to rainfall and irrigation, in the first year of the simulation can be seen in Figure 5-22. Decreases in EC coincided with rainfall events, particularly in the wet season. The increases in EC coincided with irrigation in between rainfall events.

![Figure 5-22. Soil solution salt concentration in response to rainfall.](image)

### 5.3.4.2 Potential Deep Drainage

Potential deep drainage in these simulations is controlled by hydraulic conductivity, rainfall, irrigation and evaporation. In the HYDRUS model, the changes in hydraulic conductivity in response to gypsum applications are the result of the combined impacts of decreases in sodicity increases in salt concentration and decreases in pH. The impacts of gypsum on hydraulic conductivity has been shown to result from electrolyte effects due to increases in salt concentration and the cation exchange effects due to decreases in sodicity (Loveday 1976). Although gypsum can reduce the pH of alkaline sodic soils (Nelson, Ham et al. 2001), the sensitivity of sodic soil hydraulic conductivity to changes in pH is likely to be related to clay mineralogy (Sumner 1993). In HYDRUS, pH has a strong impact on hydraulic conductivity (see section 4.6.2).
Rainfall and irrigation water in between gypsum applications would be expected to alter the soil chemistry and hence soil hydraulic conductivity. If the soils remain sodic after the gypsum treatment, the reduction in electrolyte concentration caused by heavy rainfall could significantly reduce hydraulic conductivity. Based on the McNeal functions used in HYDRUS for hydraulic conductivity reduction, when the salt concentration is less than 5 meq/L ESPs greater than 10 would result in reductions in hydraulic conductivities. Rainwater or irrigation water with low sodium adsorption ratios may decrease soil sodicity slightly due to cation exchange. When this occurs, hydraulic conductivity may increase slightly near the soil surface.

Using these simulations with time variable applied solution chemistry, it was possible to compare the effects of different frequencies of gypsum applications. In addition, the time lags involved before the effects of gypsum are observed in the deep drainage rates and the time taken for the effects of gypsum applications to wear off were also explored. Figure 5-23 shows the deep drainage rates for the cases with no gypsum, three applications of gypsum in ten years and ten applications of gypsum in ten years. The vertical lines represent the timing of the gypsum applications with the numbers at the top of each line displaying the frequency of gypsum applications included at each interval, where (3,10) indicates that gypsum was applied at this time interval for both the three times in ten years and the ten times in ten years scenarios.

Yearly gypsum applications lead to increased deep drainage rates when compared with the other treatment frequencies. For the case where gypsum was applied three times in ten years, there were two three year periods in between gypsum applications. By comparing the deep drainage rates where no gypsum was applied to the deep drainage rates where gypsum was applied three times in ten years, the time taken for the effects of a gypsum application to wear off could be determined. In this case, the deep drainage rates with three applications in three years can be seen to begin to approach the curve for no gypsum applications, in the years following the first application of gypsum. In the case where gypsum was only applied once, it took eight years before the deep drainage rates decreased back to the deep drainage rates with no gypsum applied.
The apparent time lag between gypsum applications and the increases in deep drainage rates are likely to be contributed to by the highly seasonal rainfall. Deep drainage rates did not increase immediately following the gypsum applications but instead increased dramatically several months later, when the rainfall season started (Figure 5-23). So the time lag is likely to be the result of the combination of the time taken for gypsum applications to alter soil hydraulic properties and the lower rates of irrigation when gypsum is applied, compared with the high rainfall rates that follow.

When there were no gypsum applications and evaporation was included, the deep drainage rates remained low after the initial drainage from the initially wet profile (Figure 5-24). In this case, any peaks in response to rainfall were dulled. When evaporation was added to the case with yearly gypsum applications, the deep drainage rates were reduced significantly (compared with gypsum applications and no evaporation) and the time lag between rainfall events and deep drainage peaks was increased (Figure 5-24). When only 10\% of the measured evaporation was added, deep drainage rates decreased by 5\% when compared with the case with no evaporation (Table 5-7). However, when the full 100\% of the measured evaporation was added, deep drainage rates decreased by greater than 40\% (Table 5-7).
Chapter 5  Assessing the potential impacts of sodic soil amelioration on deep drainage

Figure 5-24. Deep drainage results when evaporation was added.

Table 5-7. Total deep drainage when evaporation was added.

<table>
<thead>
<tr>
<th></th>
<th>No Evaporation</th>
<th>10% Evaporation</th>
<th>100% Evaporation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total deep drainage over 10 years (cm)</td>
<td>663 cm</td>
<td>624 cm</td>
<td>381 cm</td>
</tr>
</tbody>
</table>

5.4 HYDRUS-2D Simulations

These simulations were then extended using HYDRUS-2D/3D (Simunek, van Genuchten et al. 2006) to assess the influence of gypsum placement within the irrigation furrows. The HYDRUS (2D/3D) software package is an extension and replacement of HYDRUS-2D (Version 2.0) and SWMS_3D (Simunek, van Genuchten et al. 2008). It incorporates most of the features available in HYDRUS-1D as well as some additional features such as the spatial root distribution functions (Simunek, van Genuchten et al. 2008).
To determine the influence of furrows on deep drainage rates, HYDRUS-2D models were constructed. In order to firstly determine whether the 2D results could be compared to the previous 1D results, a model was set up in HYDRUS-2D with the same model inputs as the HYDRUS-1D model with yearly gypsum applications. As the feature for allowing water to pond on the surface was not available in HYDRUS-2D, the HYDRUS-1D model used for comparison was modified to remove this surface ponding.

HYDRUS-2D models were then set up to represent a 5.8 metre transect across an irrigated field with three furrows spaced exactly one metre apart. Simplified geometry was used where the furrows were positioned below an otherwise flat field (Figure 5-25). Automatic mesh generation was used with the target mesh size set at 30 cm. After initial models did not converge when irrigation water started to infiltrate, a mesh refinement was applied near the soil surface surrounding the furrows where infiltration occurs. The finer mesh near the surface improved model stability and lead to convergence. While the coarse mesh throughout the rest of the profile may lead to model instabilities, these simulations were only used to broadly assess the significance of gypsum placement within furrows.

By using the HYDRUS-2D model, it was possible to specify different applied solution compositions in the furrows than those between the furrows. The compositions used were rainfall between the furrows with river water plus gypsum saturated river water in the furrows. The timing for the rainfall and irrigation was kept the same as that described in section 5.3.3.1. The other model inputs were also kept the same as the 1D cases i.e. clay loam soil, seasonal upper boundary condition (including rainfall, irrigation and evaporation), free drainage lower boundary condition and hydraulic conductivity reduction due to chemistry.
5.5 HYDRUS-2D Results

To determine whether differences between the HYDRUS-1D and HYDRUS 2D/3D software lead to differences in deep drainage results when modelling sodic soil amelioration, the same model conditions were set up for both (as described in section 5.4). For the 1D model setup in HYDRUS 2D (reflected as a 1 cm wide column), the total deep drainage over a ten year period with yearly gypsum applications was 608 cm. For the equivalent model set up in HYDRUS 1D, the total deep drainage over the same period was...
663 cm. Reasons for the lower total deep drainage simulated by HYDRUS-2D are not yet clear.

While the differences in deep drainage rates predicted for HYDRUS-1D and HYDRUS-2D need to be kept in mind, HYDRUS-2D was used to explore the impacts of geometric effects due to variations in surface topography. This was achieved by setting up a cross-sectional model using HYDRUS-2D to investigate the impact of gypsum applications in furrows. For this investigation, soil sodicity profiles and deep drainage rates were compared for application of gypsum only in furrows versus gypsum application across the entire upper boundary. The furrows are present in the geometry for both cases so that this factor is kept constant.

When gypsum was applied across the entire upper surface, sodicity reductions occurred fairly uniformly in the upper parts of the soil profile, with the concentrations reflecting the surface topography (Figure 5-26). This contrasts with the sodicity profiles that resulted from gypsum applications being limited to the furrows (Figure 5-27). Note that the colour schemes are different for Figure 5-26 and Figure 5-27 due to differences in the maximum and minimum sodium concentrations between the two scenarios. Where gypsum applications were limited to the furrows, sodicity reductions were greatest directly beneath the furrows with sodicity levels increasing in between the furrows.

For making comparisons of deep drainage with gypsum applications only within the furrows or across the entire upper surface, the differences in total gypsum application rates need to be considered. The concentration of gypsum in irrigation water was kept the same but when the gypsum solution was applied only in the furrows, the gypsum application rate per hectare was reduced because gypsum was applied to a smaller area. To take this into account, the differences in deep drainage rates are considered with reference to the relative gypsum application areas.

When gypsum was applied across the entire upper surface, the total distance across the surface where gypsum was applied was approximately 629 cm. When gypsum was only applied within the furrows, the total surface length for the three furrows was 229 cm, therefore slightly more than one third of the length for gypsum applied everywhere. When
the total deep drainage after ten years was divided by the total width of the modelled cross-section, the deep drainage for gypsum applied everywhere was 546 cm whereas the deep drainage for gypsum only applied only in the furrows was 310 cm, therefore more that half of the deep drainage for gypsum applied everywhere. Applying gypsum only in the furrows resulted in more than half of the deep drainage compared with gypsum applied everywhere even though only around a third of the quantity of gypsum was applied.

Figure 5-26. Exchangeable sodium concentrations in the upper 1.5 metres of the soil profile when gypsum was applied everywhere.

Figure 5-27. Exchangeable sodium concentrations in the upper 1.5 metres of the soil profile when gypsum was only applied in the furrows.
5.6 Discussion

Using the soil water chemistry dependent hydraulic conductivity option in HYDRUS lead to model instabilities. This is partly because it is more difficult to solve the problems when the hydraulic conductivity varies throughout the profile over time. This means there are a number of cases of interest where the model could not be run, including gypsum directly incorporated into the soil profile, application of high salinity waters and seasonal boundary conditions with “clay” hydraulic properties (e.g. soils with very high percentages of clay (e.g. > 60%)). The seasonal boundary condition cases were run with “clay loam” hydraulic properties instead and this would most likely have increased the predicted deep drainage. It did, however, still allow relative comparisons between the effects on potential deep drainage for different gypsum application frequencies as well as of the influence of seasonal rainfall on soil chemistry and potential deep drainage.

When the hydraulic conductivity reduction option in HYDRUS-1D is used to estimate the impacts of sodic soil amelioration, the limitations of the functions used for this option need to be understood. The functions are specific to the soils tested by McNeal and Coleman (1966) and Suarez, Rhoades et al. (1984). With these functions, ESPs below 20 have little influence on hydraulic conductivity and small changes in pH have a large influence on hydraulic conductivity. For example, an increase in pH from 8 to 9, can cause a decrease in hydraulic conductivity of approximately sixty percent. The responses of individual soils to changes in ESP and pH may be smaller or larger than the responses included in the HYDRUS model functions and this may lead to a simulated over- or under-estimation of the impacts of sodic soil amelioration.

There were a number of other simplifications and assumptions made for the simulations discussed in this Chapter which influence the interpretation of model results. Some of the simplifications used for the seasonal boundary condition simulations include using one year of irrigation data and not incorporating crops into the simulations. Repetition of the irrigation pattern leads to similarities in deep drainage rate patterns in different years. Use of ten years of rainfall data helps to show the variation in deep drainage rates in response to large variation in inputs. Incorporation of crops and root water uptake from various
depths within the soil profile into the simulations is expected to decrease deep drainage and influence soil salinity in the root zone.

Another simplification was the use of uniform soil profiles instead of incorporating features associated with different horizons as observed in the BHWSS in previous studies (Donnollan 1991; McClurg, Tucker et al. 1988). Detailed chemical and physical soil measurements were not available for the entire depth of the unsaturated zone that was simulated. As uniform soil profiles were used, the impacts of heterogeneities in the stratigraphy or preferential flow were not considered. These factors are also expected to alter deep drainage in the field and warrant further analysis.

Some of the assumptions made relate to model parameters that were selected that were not based on measured properties. These include the soil hydraulic properties, the composition of the rainwater, the ponding depth for the seasonal boundary condition simulations and most of the solute transport parameters. The hydraulic properties used were average soil properties for a given soil type based on data from Carsel and Parrish (1988) or Schaap, Leij et al. (2001). As these inputs are likely to differ from the actual properties in the field, the model predictions can not be expected to exactly match conditions at the actual field site. Instead, the model can be used to better understand the processes involved and to determine the relative changes in response to different treatments.

There is a need to consider the environmental consequences of sodic soil management practices (Jayawardane and Chan 1994). The UNSATCHEM model, which has been incorporated into HYDRUS, has been used previously to model sodic soil reclamation in conjunction with a field study (Suarez 2001). In this UNSATCHEM modelling, the soil was initial saline and sodic so EC and chloride concentrations were used as measures of leaching of the excess salts and SAR profiles were used as measures of reclamation using gypsum (Suarez 2001).

The modelling in this Chapter was used to assess the potential impacts of sodic soil amelioration on deep drainage. Sodicity profiles were also presented to show extents of sodic soil reclamation i.e. the depth of sodicity reductions. Increases in deep drainage rates are expected to be related to the depth of sodicity reductions. This is because, the greater
Chapter 5 Assessing the potential impacts of sodic soil amelioration on deep drainage

the depth of the profile that is reclaimed, the higher the likelihood that the increases in infiltration rates will result in increases in groundwater recharge rates due to increased hydraulic conductivities further down the profile.

Some of the factors that were considered in this Chapter when assessing the impacts of sodic soil amelioration were initial soil properties, gypsum application frequency, rainfall and evaporation. The soil properties that were investigated were the initial soil sodicity and soil physical properties that influence hydraulic properties, namely bulk density and soil texture.

All of the soil properties tested were found to have a strong influence on sodic soil reclamation and deep drainage rates. There was an increase in deep drainage of greater than 60% when the clay percentage was decreased from 40 to 20 which corresponds to a change in soil type from a “loam” to a “clay”. However, for changes in soil texture within the “clay” soil range, i.e. a reduction in clay percentage from 90 to 40, deep drainage only increased by approximately 50%, indicating that for clay soils, small variations in clay percentage are not a significant control on deep drainage. When the bulk density was increased from 1.4 g/cm$^3$ to 2.0 g/cm$^3$, deep drainage decreased by an order of magnitude. This gives an indication of how much hydraulic conductivities in the field may vary on the basis of bulk density differences alone.

High clay contents and/or bulk densities in field soils in the Lower Burdekin might therefore be expected to result in reduced deep drainage. In addition, the reduced hydraulic conductivities in these soils are likely to slow down responses to sodic soil amelioration.

Deep drainage was found to be highly sensitive to the initial ESP of the soil. This was particularly the case for initial ESPs of greater than 16.9, where deep drainage was effectively non existent. Variations in the initial ESP also controlled the depth of reclamation that could be achieved. For ESPs below 16.9, most of the 10 metre soil profile was reclaimed over a 10 year period, with continuous gypsum applications. By assessing the impacts of continuous gypsum applications, cumulative effects could be determined.
In contrast, the results using a seasonal upper boundary condition show the impacts of varied gypsum application frequencies on the depth of the profile ameliorated. After one gypsum application, only the sodicity in the top 10 cm of the profile was reduced. When gypsum applications were repeated at yearly intervals, the sodicity in the top 40 cm of the profile was reduced by the end of a ten year period. Even when no gypsum was applied, there were sodicity reductions due to irrigation water applications and rainfall were observable in the model results but only in the top 5 cm of the profile. Soil salinity was also found to be reduced in response to seasonal rainfall.

Even though gypsum only decreased the sodicity in the top 40 cm of the profile over a ten year period, there were observable increases in deep drainage after only one gypsum application. The increases in deep drainage following gypsum applications did not occur immediately after the application but instead occurred immediately after seasonal rainfall. This indicates that the highly seasonal rainfall in the Lower Burdekin contributes to the impacts of gypsum applications on deep drainage.

In the Lower Burdekin, high rainfall rates are coupled with high evaporation rates, therefore the combined effects of rainfall and evaporation on deep drainage were also tested. The inclusion of evaporation in the seasonal boundary condition simulations lead to reductions in deep drainage where gypsum was applied and essentially removed deep drainage in the case where no gypsum was applied.

The impacts of different irrigation treatments, namely “river water”, “shandied water” and “gypsum dissolved in river water” were studied. All treatments were found to lead to increases in deep drainage. It is important to note that all of the treatments added additional salts to the soil compared with rainfall alone and that the “shandied” irrigation waters added the largest quantity of salts.

Using HYDRUS 2D, the possible impacts of gypsum applications within furrows could be investigated. The location of gypsum applications was found to alter the patterns of sodicity reductions. When the gypsum was only applied in the furrows, sodicity reductions occurred directly beneath the furrows but increases in sodicity occurred between the furrows. When gypsum was only applied in the furrows, deep drainage was reduced by
more than 40% compared with when gypsum was applied across the entire upper surface, however, the total quantity of gypsum applied was also reduced because the concentration of gypsum in solution was kept the same but the area to which this solution was applied was reduced. However, when gypsum was only applied in furrows, the quantity of gypsum applied was reduced by greater than 60% whereas deep drainage was only reduced by 40%. This disparity is probably the result of the contribution of rainfall to deep drainage. Although less gypsum was applied, rainfall was still applied in between the furrows and this would contribute to deep drainage even though the sodicity reductions were found to occur directly beneath the furrows, where the gypsum was applied.

The results of the modelling in this Chapter, where sodic soils were ameliorated using gypsum, followed the general trends that have been presented in published studies. Field studies have measured increased rates of water entry (Acharya and Abrol 1977; Davidson and Quirk 1961) and decreases in exchangeable sodium concentration (Acharya and Abrol 1977; Greene and Ford 1985) in response to gypsum treatments on sodic soils. When soils are successfully ameliorated, the leaching flux is expected to increase relative to surface runoff potentially increasing deep drainage rates (Rengasamy and Olsson 1993). This has also been demonstrated in field studies. For example, McIntyre, Loveday et al. (1982) measured increases in deep percolation (below 2 metres) when gypsum was applied to saline-sodic cracking clay soils. In addition, gypsum applications were predicted to lead to increased deep drainage rates under a sodic duplex soil in the Lower Burdekin based on a solute mass balance model (Dowling, Thorburn et al. 1991). The model predictions in this study support these earlier predictions.

5.7 Conclusions

There are simplifications and limitations associated with the simulations presented in this Chapter, however, some of the findings from these simulations can be used to address research questions that are applicable in the field. Saturated solutions of gypsum dissolved in river water had more effect on hydraulic conductivity than the shandied water. This was partly due to the strong dependence of hydraulic conductivity on pH. Higher salinity
shandied solutions may lead to higher hydraulic conductivities but high salinity waters would not be suitable for irrigation.

The potential impacts of gypsum on deep drainage were found to be controlled by soil properties, particularly texture, bulk density and initial exchangeable sodium concentrations. High evaporation rates lessened the impacts of gypsum on deep drainage. Decreasing the frequency of gypsum applications also lessens the impacts on deep drainage as the effects of gypsum applications on deep drainage appear to diminish after several years.

For the simulations where gypsum was applied only in furrows, sodicity levels were reduced directly beneath the furrows and actually increased in between the furrows. Applications of gypsum only within the furrows resulted in less deep drainage than application of gypsum over the entire upper surface. However, rainfall in between irrigation furrows also contributes to the total deep drainage.

Gypsum applications increased deep drainage rates even when sodicity reductions and increases in hydraulic conductivity only occurred in the upper soil profile. For the BHWSS soils with clay contents, if evapotranspiration rates were high, deep drainage rates would be reduced substantially and the effects of amelioration would be much less noticeable. The irrigation data used in these simulations was from a site where efforts were being made to improve irrigation practices. At other locations in the BHWSS, irrigation rates may be higher and this would speed up responses to soil amelioration and increase deep drainage rates.
CHAPTER 6 – CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

Groundwater tables are rising beneath irrigated fields in some areas of the Lower Burdekin in North Queensland, Australia. The soils where this occurs are predominantly sodic duplex soils and cracking clays and both of these soil types commonly exhibit high sodicity levels. Many of these soils have been treated by applying gypsum or by increasing the salinity of irrigation water by mixing saline groundwater with fresh river water. While the purpose of these treatments is to increase infiltration into the surface soils and improve productivity of the root zone, the treatments may have also contributed to deep drainage and rising water tables.

This thesis assessed the potential for amelioration of sodic soils to increase deep drainage, in the context of sodic soil management practices in the Burdekin Haughton Water Supply Scheme (BHWSS). The approach used was to collect samples of sodic soils from the BHWSS, measure changes in hydraulic conductivity for these samples in response to a representative range of amelioration treatments in the laboratory and simulate potential changes in soil profile chemistry and deep drainage using the HYDRUS model.

6.1.1 Sodic Soils in the Burdekin Haughton Water Supply Scheme

In Chapter 2, the results of pre-development soil surveys and recent soil sampling were described. Within the BHWSS, the Mulgrave area was found to have the greatest area of sodic soils. As a result of the investigations, it was found that strongly sodic soils are still present including some soils under irrigation. Sodicity levels were found to increase with depth and high sodicity levels were detected even below the root zone.
The combination of high sodicity levels and high clay contents in the soil profile coupled with clay rich materials extending up to ten metres below the soil surface might be expected to minimize the impacts of sodic soil amelioration on deep drainage. However, groundwater recharge appears to be increasing and there is a need to know to what extent the amelioration may be affecting deep drainage.

6.1.2 Laboratory Experiments

When sodic soils are ameliorated, soil hydraulic properties are altered in response to changes in soil chemistry. The expected responses of sodic soil hydraulic conductivity to these changes in soil chemistry depend on soil properties such as clay mineralogy, mineral weathering, organic matter and the presence of oxides. To determine the changes in the hydraulic conductivity of specific sodic soils in response to amelioration treatments, laboratory measurements are needed.

In Chapter 3, using repacked sodic clays sampled from the Mulgrave section of the BHWSS, column leaching experiments were performed and the impacts of gypsum and mixed cation salt solutions on hydraulic conductivity were determined. The final hydraulic conductivities measured when saturated gypsum solutions were applied were at least ten times higher than the hydraulic conductivities measured using a low salinity mixed cation solution as an irrigation water surrogate. When saturated gypsum solutions were applied, the hydraulic conductivity gradually increased to a maximum during a leaching period of at least three weeks. The increase in hydraulic conductivity was associated with a substantial reduction in soil sodicity, from an ESP of > 15 to an ESP of < 1.5.

To test the influence of bulk density on measured hydraulic conductivity, two different bulk densities were used for the column experiments where saturated gypsum solutions were applied, with five replicates for each bulk density. The hydraulic conductivity for the columns packed at 1.3 g/cm$^3$ was significantly higher, at a probability level of 0.05, than the hydraulic conductivity for the columns packed at 1.4 g/cm$^3$. The average hydraulic conductivity for the columns packed at 1.3g/cm$^3$ was 0.964 cm/day whereas the average hydraulic conductivity for the columns packed at 1.4 g/cm$^3$ was 0.147 cm/day. For each
bulk density, there was significant variation in hydraulic conductivities between replicates, but the variation between replicates was higher for the columns packed at 1.3 g/cm$^3$. For both bulk densities, higher final hydraulic conductivities were found to correlate with lower final exchangeable magnesium concentrations. This indicates that for these soils, calcium increases hydraulic conductivity more than magnesium.

The trends in hydraulic conductivity when mixed cation salt solutions were applied consisted of an increase to a maximum hydraulic conductivity during the first 2 to 4 weeks, followed by a very gradual decrease to an equilibrium value. The maximum hydraulic conductivities for the 100 meq salt solution were approximately double the maximum hydraulic conductivity for the 50 meq salt solution. For this range of salt concentrations, the increase in hydraulic conductivity presented in this thesis was significantly greater than the increases found by McNeal and Coleman (1966) for a large range of soil sodicity levels. The time taken to reach the final stable hydraulic conductivity was longer for the 100 meq solution than for the 50 meq solution but the number of pore volumes that passed through the soil columns was almost the same.

The observed trends of a sharp increase in hydraulic conductivity to a maximum followed by a gradual decrease in hydraulic conductivity with a mixed cation salt solution treatment have not been previously reported in the literature, to the best of my knowledge. There are a number of possible reasons for the initial increase in hydraulic conductivity, including decreases in soil sodicity, decreases in pH and enhanced electrolyte effects due to salt dissolution. A theory was presented for the slow decrease in hydraulic conductivity, namely that the clay minerals are undergoing diagenesis due to long periods of saturation with the saline mixed cation solutions. Measurements of the clay mineralogy of the soil used in this study indicate that changes in clay mineralogy may have taken place, with the proportion of smectite and illite lower than expected, and the proportion of kaolinite higher than expected. However, the alteration of clay minerals is considered as long term process and these changes in clay mineralogy are not expected to occur in an irrigated field in the absence of long term saturation with saline waters.
6.1.3 Modelling

Chapter 4 assessed the applicability of the HYDRUS-1D model to the simulation of sodic soil amelioration. The HYDRUS-1D model contains a function which relates the changes in soil chemistry to changes in hydraulic conductivity, as originally incorporated in the UNSATCHEM model (Simunek, Suarez et al. 1996). However, this function should be applied with caution as the responses of individual soils to changes in soil chemistry may vary from the responses given by this generic function.

The UNSATCHEM simulations of Simunek and Suarez (1997) were used as a starting point for modelling the laboratory experiments of Chapter 3. Due to differences in flow modelling between UNSATCHEM and HYDRUS-1D, some of the published results, particularly those with acidic solutions applied, could not be reproduced. However, modelling of gypsum treatments in HYDRUS-1D lead to a satisfactory match with the published UNSATCHEM results. As gypsum was the primary sodic soil treatment considered in this thesis, the reproduction of the results for gypsum treatments was sufficient for confirming the applicability of HYDRUS-1D for the simulation of sodic soil amelioration.

Modelling of the column experiments confirmed that the overall trends and key chemical reactions could be adequately simulated. For the experiments where gypsum was applied, the model results showed gradual increases in hydraulic conductivity occurring at the same time as decreases in soil sodicity, as occurred in the laboratory.

In Chapter 5, different amelioration treatments were tested and gypsum dissolved in river water was found to result in a greater increase in hydraulic conductivity than the “shandied water”, a mixture of saline groundwater with river water. This was partly due to the higher pH for the shandied water and the strong dependence of hydraulic conductivity on pH in HYDRUS-1D. Higher salinity shandied solutions can lead to higher hydraulic conductivities but high salinity waters, e.g. salinities above 1500 µs/cm, would not be suitable for sugarcane irrigation. In the simulations where gypsum was applied at different frequencies, gypsum applications increased deep drainage rates even when sodicity reductions and increases in hydraulic conductivity only occurred in the upper soil profile.
While gypsum applications were found to lead to a potential increase in deep drainage, the actual impacts of gypsum on deep drainage were found to be influenced by clay content, bulk density, initial exchangeable sodium concentrations and the frequency of gypsum applications. Increases in any of these soil properties lead to reductions in deep drainage and delayed the impacts of gypsum on deep drainage. When the frequency of gypsum applications was reduced the effects of gypsum applications on deep drainage were shown to diminish after several years.

The increases in deep drainage did not occur immediately after each application of gypsum, but instead occurred a few months later, immediately after seasonal rainfall. This indicates that the highly seasonal rainfall in the Lower Burdekin contributes to the impacts of gypsum applications on deep drainage.

The simulation results using HYDRUS-2D showed that when gypsum was applied only in the furrows, sodicity was reduced directly beneath the furrows while sodicity levels actually increased between the furrows due to displacement of sodium from directly beneath the furrows. When sugarcane crops are planted in the rows between the furrows, these increases in soil sodicity in between the furrows may adversely impact upon water uptake by roots due to local decreases in hydraulic conductivity.

When gypsum was only applied in furrows, approximately one third of the quantity of gypsum was applied compared with when gypsum was applied across the entire soil surface. The deep drainage resulting from gypsum applications only within the furrows was approximately half of the deep drainage resulting from gypsum applications across the entire soil surface. Therefore, the quantity of deep drainage relative to the quantity of gypsum applied was higher when gypsum was only applied in furrows than when gypsum was applied to the entire soil surface. However, there are also other factors that contributed to deep drainage in these simulations. Rainfall, which falls across the entire soil surface, contributed to deep drainage regardless of where the gypsum was applied.
6.1.4 Implications for Amelioration of Sodic Soils in the Lower Burdekin

This thesis has addressed the potential for sodic soil amelioration, which impacts on the soil chemistry, to increase hydraulic conductivity and deep drainage. Based on the results of the column experiments, there is potential for large increases in hydraulic conductivity in response to gypsum or increased salinity of irrigation waters. The increases in hydraulic conductivity that occurred during the column experiments occurred with continuous application of the treatments.

Whether increases in groundwater recharge will occur in response to these increases in hydraulic conductivity depends on local factors including in situ soil physical and chemical properties including soil structure and heterogeneities, the depth of the upper clay layer, characteristics of the actual amelioration treatment chosen, irrigation, rainfall and evaporation. HYDRUS modelling results for a ten metre depth sodic clay soil showed that increases in hydraulic conductivity resulting from gypsum applications contributed to increased deep drainage. When 7 tonne/hectare annual gypsum applications were simulated over a 10 year period with seasonal irrigation and rainfall, reductions in sodicity occurred in the top 50 cm of the soil profile that resulted in increased water fluxes to this depth, and corresponding increases in deep drainage below the 10 m profile simulated. When only one 7 tonne/hectare gypsum application was simulated at the start of a ten year simulation period, reductions in sodicity occurred in the top 10 cm of the soil profile. Even with this shallow depth of sodicity reductions, deep drainage rates were increased. However, during the ten year period simulated, deep drainage rates decreased again when no further gypsum was applied. This means that increases in deep drainage expected in the field would be proportional to the quantity of gypsum applied.

One of the findings from the modelling that is applicable to the field setting was that the gypsum dissolved in river water had a greater impact on hydraulic conductivity than the “shandied” irrigation water. While gypsum resulted in 98% reduction in exchangeable sodium concentrations at a depth of 1 metre after 10 years, the “shandied” irrigation water only resulted in a 31% reduction in exchangeable sodium concentrations at a depth of 1 metre after the same time period. The comparison between different treatments was based on continuous application of each treatment. Gypsum would result in more significant
impacts on hydraulic conductivity than shandied water due to the soil sodicity reductions resulting from gypsum applications. As a consequence, gypsum would also result in higher deep drainage than “shandied” irrigation water.

Rainfall and irrigation water in between gypsum applications reduce the effects of the gypsum by decreasing the electrical conductivity of the soil solution. In the same way, flushing by rainfall in between applications of high salinity water would lessen the effects of this treatment. Rainfall can cause significant decreases in the hydraulic conductivity of soils treated with high salinity waters because the soil sodicity would remain high where gypsum is not applied. In contrast, where soil sodicity is reduced and hydraulic conductivities are increased by sodic soil amelioration, heavy rainfall in the wet season would actually result in a greater contribution than irrigation to deep drainage in the area.

Variations in evapotranspiration, rainfall and irrigation rates in the field have a significant influence on deep drainage rates. For the BHWSS soils with high clay contents, if evapotranspiration rates were high, deep drainage rates would be reduced substantially and the effects of amelioration would be much less noticeable. The irrigation data used in these simulations was from a site where efforts were being made to improve irrigation practices. At other locations in the BHWSS, irrigation rates may be higher and this would speed up responses to soil amelioration and increase deep drainage rates. Increases in rainfall would also lead to increased deep drainage rates.

The most important finding is that in order to predict deep drainage rates in the Lower Burdekin, the influence of amelioration on soil chemistry and hydraulic conductivity needs to be considered. Under traditional modelling approaches, saturated hydraulic conductivity is not normally considered to be a dynamic soil property and the influence of gypsum and/or varied irrigation water quality on deep drainage is not considered. The laboratory and modelling results in this thesis have demonstrated that the hydraulic conductivity of sodic clay soils in the BHWSS is a dynamic property, highly dependent on soil chemistry for sodic soils. This also highlights the need for accounting for changes in water quality when modelling water fluxes in irrigated areas.
6.2 Recommendations for Further Research

Recommendations for further research based on the findings of the laboratory component of this thesis include:

(i) investigations into whether biological activity is causing or contributing to pore clogging during long term leaching

(ii) investigations into whether changes in soil structure are occurring during column experiments and relocation of clays is causing or contributing to pore clogging during long term leaching

(iii) further column experiments are required to study the effects of gypsum and irrigation water salinity on soil hydraulic conductivity which cover a larger range of amelioration treatments and which incorporate investigations of wetting and drying cycles

(iv) extension of the laboratory experiments using intact cores. Preliminary tests with intact cores in this thesis indicate that large gradients were needed to obtain flow. Use of intact cores that are much larger than the repacked columns used in this study would capture more of the field heterogeneity and improve representativeness of the laboratory measurements. Using cores from multiple depths in the soil profile would also provide more information on the type of variability encountered within the root zone.

Investigation into possible causes for pore clogging during long term leaching may help to explain the reasons for the decline in hydraulic conductivity observed when sodic clays where leached with mixed cation solutions. Extension of the column experiments to cover a larger range of amelioration treatments and conditions and the use of intact cores would improve the applicability of the laboratory results to field conditions.

Recommendations for further research based on the HYDRUS modelling carried out in this thesis include:

(i) improvements to the HYDRUS model code to improve its ability to handle the simulation of fine textured sodic clay soils when the hydraulic conductivity reduction function is used
(ii) simulations using longer time periods of records for the irrigation data and measured rainfall composition. This may improve the applicability of the models to studying the effects of sodic soil amelioration.

(iii) investigations into alternative functions relating soil chemistry to hydraulic conductivity that are more widely applicable to sodic soils, even if soil specific parameters are still needed for these functions.

Future modelling of soils and groundwater in the Lower Burdekin will require consideration of the influence of soil chemistry on soil hydraulic conductivity to further improve predictions of deep drainage and groundwater fluxes. As crop growth models will be needed to adequately simulate the water and solute fluxes within the root zone, cropping systems models will need to include dynamic soil chemistry dependent hydraulic conductivity so that all key processes relevant to predicting deep drainage are included.
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APPENDIX
Model setup information for the Simunek and Suarez (1997) simulations in Chapter 4.

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<td>Upper Optimal Iteration Range</td>
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### Solute Transport

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#### Solute Composition

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#### Adsorption Concentrations

Ca = 40, Mg = 40, Na = 120, K = 0

#### Chemical Parameters

**Kinetic Precipitation/Dissolution**

- Critical Ionic Strength (Debye-Huckel*Pitzer equations) 0
- Maximum Number of Iterations 1

### Solute Transport and Reaction Parameters

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<td>Dolom. SA</td>
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<td>K[Ca/Mg]</td>
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<td>K[Ca/Na]</td>
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### Solute Transport Boundary Conditions

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<td>CO₂ - Bottom of profile</td>
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**Carbon Dioxide Model Parameters**

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<td>Molecular diffusion of CO₂ in water</td>
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<td>Longitudinal dispersivity for CO₂</td>
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**Carbon Dioxide Production - Soil Microorganisms**

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<tr>
<td>Michaelis’ constant</td>
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<tr>
<td>Optimal pressure head</td>
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<td>Salinity stress coefficient h50</td>
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**Carbon Dioxide Production - Plant Roots**

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<td>Michaelis’ constant</td>
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<td>Degree Days to Maximum Production</td>
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*The figures in italics were only used for the reclamation with acid water at pH 2.05.*