TEMPORAL SODIUM FLUX IN A WOODLOT SOIL
IRRIGATED WITH SECONDARY TREATED EFFLUENT:
THE IMPLICATIONS FOR SUSTAINABLE IRRIGATION
AND SOIL MANAGEMENT

by

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The University of Newcastle, NSW, Australia
CERTIFICATION

I, Steven Andrew Lucas, certify that the substance of this thesis has not been submitted
for any degrees and is not currently being submitted for any other degree or
qualification. I certify that any assistance received in preparing this thesis, and all
sources used, have been acknowledged in this thesis.

…………………………………..     …………………..

Steven Andrew Lucas      Date
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ABBREVIATIONS, ACRONYMS AND SYMBOLS

<table>
<thead>
<tr>
<th>Abbreviation</th>
<th>Term</th>
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<tbody>
<tr>
<td>Δ</td>
<td>symbol prefix for “change in “</td>
</tr>
<tr>
<td>θ</td>
<td>volumetric soil moisture</td>
</tr>
<tr>
<td>θg</td>
<td>gravimetric soil moisture</td>
</tr>
<tr>
<td>cmol(+)/kg</td>
<td>centimole per kilogram pertaining to cationic charge</td>
</tr>
<tr>
<td>C&lt;sub&gt;TH&lt;/sub&gt;</td>
<td>threshold concentration</td>
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<tr>
<td>C&lt;sub&gt;TU&lt;/sub&gt;</td>
<td>turbidity concentration</td>
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<tr>
<td>CEC</td>
<td>cation exchange capacity</td>
</tr>
<tr>
<td>CI</td>
<td>confidence interval</td>
</tr>
<tr>
<td>C</td>
<td>final concentration (solubility curves)</td>
</tr>
<tr>
<td>Co</td>
<td>initial concentration (solubility curves)</td>
</tr>
<tr>
<td>D</td>
<td>deep drainage</td>
</tr>
<tr>
<td>DDL</td>
<td>diffuse double layer theory</td>
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<tr>
<td>EAT</td>
<td>Emerson Aggregate Test</td>
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<tr>
<td>EC</td>
<td>electrical conductivity (dS/m)</td>
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<td>EP</td>
<td>equivalent populations</td>
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<tr>
<td>ESP</td>
<td>Exchangeable Sodium Percentage</td>
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<td>ΔESP</td>
<td>change in soil ESP</td>
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<tr>
<td>ET</td>
<td>evapotranspiration</td>
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<tr>
<td>FC</td>
<td>field capacity</td>
</tr>
<tr>
<td>ΔI</td>
<td>irrigation surplus/deficit</td>
</tr>
<tr>
<td>ΔCI</td>
<td>cumulative irrigation surplus/deficit</td>
</tr>
<tr>
<td>ICP-AES</td>
<td>inductively coupled plasma atomic emission spectrometer</td>
</tr>
<tr>
<td>IL</td>
<td>interception loss</td>
</tr>
<tr>
<td>K&lt;sub&gt;sat&lt;/sub&gt;</td>
<td>saturated hydraulic conductivity</td>
</tr>
<tr>
<td>meq/L</td>
<td>milliequivalent per litre</td>
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<tr>
<td>mmol/L</td>
<td>millimole per litre</td>
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<tr>
<td>OS</td>
<td>outside solution</td>
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<tr>
<td>PET</td>
<td>potential evapotranspiration</td>
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<tr>
<td>Abbreviation</td>
<td>Description</td>
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<td>--------------</td>
<td>--------------------------------------------------</td>
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<tr>
<td>PVC</td>
<td>polyvinylchloride</td>
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<tr>
<td>Q_p</td>
<td>precipitation</td>
</tr>
<tr>
<td>Q_e</td>
<td>applied effluent</td>
</tr>
<tr>
<td>R</td>
<td>runoff</td>
</tr>
<tr>
<td>R_d</td>
<td>retardation factor (solubility curves)</td>
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<tr>
<td>RSD</td>
<td>relative standard deviation</td>
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<tr>
<td>SAR</td>
<td>Sodium Adsorption Ratio</td>
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<tr>
<td>SAR_p</td>
<td>Sodium Adsorption Ratio for soil in 1:5 distilled water</td>
</tr>
<tr>
<td>STE</td>
<td>secondary treated effluent</td>
</tr>
<tr>
<td>WP</td>
<td>wilting point</td>
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<tr>
<td>WWTW</td>
<td>wastewater treatment works</td>
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ABSTRACT

This study reports results obtained and the approach taken in investigating the temporal sodium flux in a woodlot soil receiving secondary treated effluent at Branxton, NSW. Previous research has shown woodlot soils receiving secondary treated effluent undergo an increase in exchangeable sodium percentage (ESP) over time. Increased soil ESP influences micro-aggregate/soil pore stability and, particularly when subject to irrigation waters of specific low-electrolyte concentrations, results in decreased soil permeability and a subsequent need to reduce effluent application rates.

Therefore, in irrigated woodlot soils it has been necessary to implement strategies to remove excess sodium from the root zone to maintain optimum permeability of the receiving soil, that is, maintaining the cation balance (as soil ESP) to promote optimum soil pore size. To maintain optimum permeability, an understanding is needed of temporal variations in the accumulation/leaching (flux) of sodium within a soil under secondary treated effluent irrigated conditions. The ability to define the sodium flux depends on the frequency of soil sampling and the ability to interpret the net loss/gain in soil sodium in relation to the applied hydraulic load over time. Past research has measured changes in soil ESP on an annual basis, or longer, making it impossible to interpret temporal sodium flux within a given year.

The rate of change of soil ESP has ramifications for optimum permeability within an effluent irrigated woodlot. With respect to increasing/decreasing soil ESP, a major response of the clay particles within micro-aggregates is the deformation of conducting soil pores and reduced hydraulic conductivities. In addition, clay dispersion is governed by the soil ESP and electrolyte concentration of the infiltrating waters at the time, where dispersed clay particles may block conducting soil pores and further reduce hydraulic conductivity. Therefore, investigating the temporal sodium flux in conjunction with the temporal variation in electrolyte concentration of infiltrating waters will give greater insight into the response of effluent irrigated soils to sodium-rich waters over time.
Three research aims were formed to investigate temporal sodium flux. These include:

1. To investigate trends in the dominant water balance components for a woodlot soil receiving secondary treated effluent (STE);
2. To examine temporal and spatial variation in both the water balance components and measured soil properties, particularly the sodium flux; and
3. To investigate the implications of the sodium flux on the loss of soil structure and drainage over time (dispersion events), particularly in relation to temporal changes in soil ESP and effluent SAR.

Monitoring programs for water balance components and soil parameters covered the period January 2002 – October 2003. Every two months, soil samples were taken at designated sites and at different depths (10, 20, 40, 60, and 80 cm). These samples were analysed for exchangeable cations (Ca\(^{2+}\), Mg\(^{2+}\), Na\(^{+}\) and K\(^{+}\)), from which the ESP values were derived. Also, this appears to be the first time that soil sampling at this frequency, which enables the temporal sodium flux to be determined, has been carried out.

Column leaching experiments were also performed over the study period to illustrate the response of the woodlot soil, in terms of micro-aggregate stability, to hydraulic loads of varying SAR. Column leaching experiments also confirmed the rate of solute movement through the soil profile and the woodlot soil’s ability to bind/exchange sodium under different hydraulic loads and electrolyte concentrations. Soil extraction plate methods were used to determine wilting point and field capacity for these soils.

The Sodium Adsorption Ratio (SAR), which is the solutional equivalent to soil ESP, was used to define the electrolyte concentration of the applied effluent and rainfall to the woodlot. The net loss/gain of exchangeable sodium (ΔESP) at different depths and times was determined and compared with changes in water balance components and the measured volumetric soil water (θ) over time. The soil water surplus/deficit was recorded at a daily time-step and a cumulative approach was used to determine the
long-term soil water surplus/deficit. In addition, variations in groundwater levels were monitored to observe if surplus irrigation events were reflected in temporal trends.

As a result of determining the temporal variation in soil ESP, effluent and rainfall SAR, daily soil water deficit/surplus (short-term), cumulative soil water deficit/surplus (long-term) and volumetric soil moisture, temporal trends are presented. The sodium flux was then investigated by interpreting trends in the monitored data with respect to the dominant water balance components. All parameters were then used to model the potential dispersive behaviour of the receiving soil over time and depth, in relation to the volume and electrolyte concentration of the effluent and rainfall applied over time. The implications for soil structure and permeability depend on variations in soil ESP and effluent SAR.

Results from this research show that soil ESP varied by as much as 24% over a four-month period and is shown to be a function of the sodium loading (from STE) and soil water surplus/deficit. On each sampling occasion, soil ESP generally increased with depth at all irrigated sites. Soil ESP at non-irrigated sites was much lower than irrigated sites, although the variability in soil ESP was much greater. Variations in SAR of the waters received by the woodlot soil (effluent and rainfall) over the study period ranged from 0.5 to 5.9. It is shown that the SAR range, coupled with variations in soil ESP, has ramifications for maintaining long-term soil structure. Soil structure at different sites within a woodlot will respond differently according to the soil ESP/effluent SAR relationship.

The dispersive potential of soil at a given ESP receiving irrigation waters of known SAR was assessed in light of the relationship between soil ESP and effluent SAR. This showed the dynamic response of effluent irrigated soils to the long-term temporal variation in electrolyte concentration of rainfall/effluent. The relationship between soil ESP and effluent SAR is graphically presented as a continuum, which in turn can be used as a management tool for assessing the potential for dispersion of clay particles in a soil of known ESP and irrigated with waters of known SAR. By identifying trends in the temporal sodium flux, the optimum permeability of the receiving soil can be assessed in relation to the electrolyte concentration of the applied waters and the soil exchangeable sodium percentage (ESP).
Secondary treated effluent application rates can then be corrected to prevent “dispersive” irrigation events over the long term and/or management strategies applied to remove excess sodium from the soil profile. The significance of the research is that a better understanding of the temporal dynamics of sodium in the soil profile will allow improved management of effluent irrigated woodlots, with the aim of making the practice sustainable with respect to controlling accumulating soil sodium and maintaining soil structure for future landuse.
Chapter 1: INTRODUCTION

The land application of secondary treated effluent (STE) is an alternative to discharging into natural waterways where environmental impacts are likely. One reuse option, which is being explored in Australia (Boardman et al., 1996; Moss et al., 1998; Myers et al., 1999) and overseas, is the growing of hardwood plantations. Irrigated hardwood trees are useful in decreasing effluent discharge to natural waterways in that they produce large amounts of biomass that not only act as a valuable carbon sink, but that they also utilise the nutrients in effluent, namely phosphorus and nitrogen. However, sodium (Na\(^+\)) is not utilised by the trees and has been shown to accumulate over time to the detriment of soil physical properties, altering micro-aggregate/soil pore stability and, as a result, decreasing permeability (Bond, 1998; Myers et al., 1999; Halliwell et al., 2001).

Deleterious structural effects are more pronounced in soils containing clay because the charged clay minerals in soils are characterised by swelling and cation exchange (McBride, 1994). Both swelling and cation exchange are dependent on the Exchangeable Sodium Percentage (ESP) of the receiving soil and the electrolyte concentration of the soil solution. At critical electrolyte concentrations, clay minerals disperse and alter the continuity of soil macropores, which consequently influences permeability (Quirk, 2001). These critical electrolyte concentrations were previously found to be applicable for a range of Australian soils, including non-arid sandy loams (Quirk and Schofield, 1955), red-brown earths (Rengasamy et al., 1984) and arid heavy clays (Quirk and Davidson, 1959).

At the critical electrolyte concentration defined as the turbidity concentration (C\(_{TU}\)), micro-aggregate breakdown and dispersion occurs and the permeability of the receiving soil decreases (Quirk, 2001). Decreased permeability occurs when clay minerals swell and deform soil pore structural stability, and/or disperse and “clog” pore spaces immediately below the dispersion zone, which may reduce permeability to zero (McNeal et al., 1968; Meneer et al., 2001; Quirk, 2001). Over time, this has ramifications for the land application of all effluents with sodium contents significantly higher than rainwater.
When secondary treated effluent is applied to irrigated woodlots, the concentration of sodium in the soil increases over time (Bond, 1998; Myers et al., 1999; Halliwell et al., 2001). However, the response of soil structure will be a function of the relative concentrations of other cations present and the relative availability of exchange sites. Increased sodium ($\text{Na}^+$) within the profile affects soil structure by swelling and ion-exchange reactions, where excess sodium can displace other cations (mainly $\text{Ca}^{2+}$ and $\text{Mg}^{2+}$) at soil exchange sites, changing the charge density and microporosity of clay components (Aylmore and Quirk, 1962, 1967; McBride, 1994).

Since permeability to air and water depends on the continuity of soil macropores (>30 µm), swollen and/or dispersed clay minerals (~20-200 µm) disrupt that continuity and, hence, the relative permeability of the soil (Oster and Shainberg, 2001). Given that maximum drainage occurs when the soil is saturated, soil saturation at optimum permeability is the optimal site condition for sodium leaching. Therefore, managing soil saturation is important in managing exchangeable soil sodium.

Optimum permeability occurs when the applied waters are electrolytically suitable to maintain micro-aggregate/soil pore stability, with respect to critical electrolyte concentrations that promote dispersion. Optimum irrigation scheduling occurs when zero surface runoff is observed, daily potential evapotranspiration is satisfied and stable groundwater levels occur. Adequate long-term soil permeability is a pre-requisite for a sustainable effluent irrigation scheme. Increased soil sodium over time has the potential to alter soil permeability when exposed to irrigation waters of specific electrolyte concentration. Therefore, managing soil sodium is related to managing long-term soil structure.

Sodium is assumed to accumulate when the soil is not saturated due to an increased soil matrix potential created by tree uptake and unsaturated soil below the root zone. However, sodium will intermittently translocate during successive wetting fronts and to a lesser extent by hydrodynamic dispersion through wetted soil pores. Wetting fronts occur when the soil dries out and is re-wetted by effluent irrigation and/or rainfall. As a result, soil ESP varies with time and depth, meaning that relative permeability also varies with time and depth. Therefore, dispersive irrigation and/or rainfall events may
reduce the capacity of a woodlot soil to receive the hydraulic load at the designed rate, due to decreases in permeability, ultimately affecting the sustainability of the practice.

Based on soil ESP, loss of soil structure would be expected whenever the sodium adsorption ratio (SAR) of the hydraulic load is less than the specific electrolyte concentration that causes dispersion, or more specifically, causes change in micro-aggregate/soil pore stability and the potential for instantaneous dispersion. The physiochemical response of soil structure to specific electrolyte concentrations, to be observed and monitored in this research, was based on the work of Quirk and Schofield (1955).

Three research aims were formed to investigate temporal sodium flux and obtain data to investigate the research questions. These include:

1. To investigate trends in irrigation scheduling for a woodlot soil receiving STE;
2. To examine temporal and spatial variation in several water balance components and measured soil properties due to irrigation scheduling, particularly the sodium flux; and
3. To investigate the implications of the sodium flux on the loss of soil structure over time, particularly in relation to temporal changes in soil ESP and effluent SAR.

For this research, the influence of short-term variation in soil ESP (months) for an effluent irrigated woodlot soil being irrigated with waters of varying SAR is shown to be important with respect to managing long-term soil structure (years). Past research has given little insight into the impact of short-term sodium flux and may have missed the potential benefits for controlling soil sodium and managing long-term soil structure. To resolve the research questions, monitored data and soil profiles of effluent irrigated sites within a woodlot were compared to monitored site data and soil profiles of non-irrigated sites, which were located immediately adjacent to the woodlot.

The temporal accumulation/leaching (flux) of sodium in effluent irrigated soils is a dynamic system influenced by many factors. This study holistically explores various components of an effluent irrigated woodlot “system” that influences temporal sodium
flux in the soil profile. These components, monitored over time, include the daily irrigation surplus/deficit (ΔI) and cumulative irrigation surplus/deficit (ΔCI) in context of several site-specific water balance components; soil ESP, volumetric soil moisture (θ), the sodium adsorption ratio (SAR) of applied effluent and/or rainfall, and groundwater, including its chemistry, direction and depth. Clay mineralogy for the studied soils were also determined to highlight specific micro-aggregate/soil pore responses to either calcium (Ca$^{2+}$) and sodium (Na$^+$), as different clay minerals display a selectivity for either calcium or sodium under different conditions (Kopittke et al., 2005).

1.1 Thesis approach and structure

Chapter 1 introduces the main topics to be discussed, the aims of this research and the structure of this thesis. Chapter 2 discusses in detail the background to this study. Included in Chapter 2 are a review of the impact of sodium in the soil profile, the significance of electrolyte concentration “thresholds” and their influence on clay particle behaviour and a discussion of the principles of effluent irrigation of woodlots.

Chapter 3 provides a detailed description of the Branxton Waste Water Treatment Works (WWTW) and its treatment processes, the effluent irrigation site and a brief overview of past monitoring data collected by the site managers, Hunter Water Corporation (HWC).

Chapter 4 describes the methods used to acquire data and the purpose of each component in relation to achieving the aims of this thesis. The first part of this chapter covers the site-specific water balance methods, while the second part covers the soil component methods, including how the behaviour of the soil profile irrigated with effluent and rainfall of differing SAR values was determined. This was achieved by utilising established theory (discussed in Chapter 2) and intensive field site soil sampling/analysis to determine temporal sodium flux over time. In addition, column-leaching experiments were performed in the laboratory, whilst soil moisture curves were measured for soils from the Branxton site. This work was undertaken to understand more about the soil by observing changes in soil structure, solute movement properties, field capacity and wilting point for those soils.
Chapter 5 presents the results of the studies and describes the temporal trends in each monitored parameter in relation to their impact on the soil sodium flux. Summary graphs (parameter versus time) are presented that describe the soil physical/chemical characteristics and effluent chemical characteristics in the context of daily irrigation scheduling. These graphs are then used to validate the proxy-indicators required to develop a conceptual irrigation scheduling framework, which aims at reducing the long-term deleterious impacts on soil structure as a result of sodium accumulation and leaching (flux) in an effluent irrigated woodlot receiving secondary treated effluent.

Chapter 6 discusses results from Chapter 5. Trends in sodium accumulation/leaching are highlighted in relation to site monitoring results and, using past theory and laboratory methods to mimic field conditions, the sodium flux for the study site was determined. The importance of daily variations in the irrigation surplus/deficit (ΔI) and subsequent variations in cumulative irrigation surplus/deficit are highlighted in this chapter, as these were deemed limiting factors for irrigation scheduling.

The development of the soil ESP/effluent SAR continuum for effluent irrigated soils as used in this research is discussed and presented graphically. Recognition and definition of proxy indicators used in the development of the continuum are also discussed. The limitations of the data acquired are then discussed in terms of the approach taken, the long-term precision/accuracy of monitoring and field equipment, and the validity of several assumptions made during this study.

Chapter 7 both summarises and concludes this thesis by reviewing the major outcomes of this research. Results from this study emphasise a need to frequently determine soil ESP and the electrolyte concentration of application waters, to assess relative permeability with the aim of maintaining optimum permeability over time in an STE irrigated woodlot. Future research direction and management options are also discussed, particularly with respect to maintaining optimum permeability.
Chapter 2: BACKGROUND

2.1 Introduction

This chapter describes several relevant theories and concepts and is presented in two stages, namely the physical/chemical behaviour of soil in relation to increasing sodium (Section 2.2) and the principles of effluent irrigation and implications for soil sodium flux (Section 2.3). Section 2.2 broadly discusses the impact of increasing soil sodium on soil structure, micro-aggregate stability, soil permeability and the relationship between soil ESP and effluent SAR. Section 2.3 ascertains the issue of increasing sodium in the soil profile due to irrigation with STE, namely a decrease in permeability over time and subsequent reductions in effluent application.

2.2 Physical and chemical behaviour of soils increasing in sodium

2.2.1 Salinity and sodicity

Soil salinity and sodicity are major environmental problems in Australia and both can be exacerbated by effluent disposal to land (Bond, 1998). Soil salinity is a characteristic of soils that relates to their content of water-soluble salts (Charman and Murphy, 2000). Typically, inorganic salts are crystalline ionic compounds that dissociate in water to form cations (\(\text{Ca}^{2+}, \text{Mg}^{2+}, \text{Na}^+, \text{K}^+, \text{Al}^{3+}, \text{Cu}^{2+}, \text{Fe}^{2+}, \text{Zn}^{2+}\)) and anions (\(\text{Cl}^-, \text{NO}_3^-, \text{PO}_4^{3-}, \text{SO}_4^{2-}, \text{CO}_3^{2-}, \text{KO}_3\)) (McBride, 1994; Patterson, 1994; White, 1997). The proportion of cations and anions in natural soil water is a product of soil type, climate and land use (Patterson, 2001). Other solutes, such as organic products, may be present but do not dissociate into ions. However, excess \(\text{Na}^+\) occupying soil exchange sites has been shown to be problematic if not properly monitored.

Soil salinity may be caused by vegetation clearance, rising groundwater tables and/or the release of marine salts from geologic parent materials (Taylor, 1996) and therefore may comprise of many different cations and anions. In the case of effluent irrigation, additional sodium is supplied to wastewater through washing powders and detergents (Patterson, 1997). As a result of excess sodium in the effluent, sodium may accumulate in soils where irrigation of effluent is undertaken (Rengasamy and Olsson, 1993; Sumner, 1993; Magesan et al., 1999).
When sodium is the dominant cation in the soil matrix the term sodicity is used. Sodic soils would most likely exhibit sodium induced problems such as hard-setting and increased dispersion and erosion. The cause of soil sodicity relates to the chemical characteristics of the effluent being applied and the ability of the soil to preferentially exchange and leach Na\(^+\) under changing soil moisture conditions (Myers et al., 1999). The ability of the soil to allow transmission through to groundwater will vary depending on soil type and the extent of micro-aggregate instability and soil pore deformation.

The need to better characterise soil response to salinity/sodicity is summarised in Oster and Shainberg (2001, p1219), who cite from El-Ashry and Duda (1999):

“There is an increasing obligation to better characterise the response of field soils to salinity and sodicity, and the effect of management on these responses. This is particularly the case for irrigated agriculture, which must increase production with less water, in many instances with higher salinity and sodicity, and at the same time reduce and controlling negative environmental impacts on surface and groundwaters.”

Since the soil profile transmits surface water to groundwater, the impacts of irrigation water quality on soil micro-aggregate/soil pore stability need to be considered when designing an irrigation schedule. The irrigated soils’ response to variations in irrigation water quality may result in reduced permeabilities at specific soil ESP values (Rengasamy and Olsson, 1993; Quirk, 1994; Oster and Shainberg, 2001), and as a consequence, poorly managed effluent irrigated soils may experience deleterious structural impacts that excludes future agricultural use of the land.

In the context of clay particle stability governing soil pore structure, excess sodium relative to calcium and magnesium will decrease downward water movement by altering optimum transmission within a soil profile. Therefore, there is a need to better understand temporal soil sodium flux in a secondary treated effluent irrigated woodlot, in order to predict changes in soil structure over time to maintain optimum water transmission through the soil.
2.2.2 Sodium in the soil profile

Effluent applied to woodlots usually comprises many cations and anions, however by occupying exchange sites previously held by other cations (eg, Ca$^{2+}$ and Mg$^{2+}$) the presence of excess Na$^+$ causes changes to the arrangements of clay particles, which may lead to significant decreases in soil permeability. Sodium chloride and other salts which are added to the soil during effluent irrigation have a propensity to accumulate in the soil as evapotranspiration proceeds (Myers et al., 1999). Additional irrigation may be required to establish a net downward flow in order to remove excess sodium (Donlon and Surapeneni, 2000), although it is commonly accepted that any effluent irrigation scheme is rarely implemented with such precision (Bond, 1998). For example, erroneous estimates of the site water balance, such as transpiration rates, effluent/rainfall interception estimates and variable and/or unpredictable rainfall will inevitably result in an irrigation deficit or surplus. Consequently, this is when environmental problems usually occur.

The amount of sodium accumulating in the soil root zone will be a factor of climate, site topography, volumetric soil water content ($\theta$) (more to dissolve and transport Na$^+$ out of the root zone or less to promote evaporative concentration), the direction of water movement in the root zone (deep drainage, capillary rise/tree water use or lateral movement), the Na$^+$ loading applied to the soil, and the exchangeable cationic inputs to soil water solution from the soil itself (Shaw, 1994; White, 1997). Theoretically, the excess exchangeable Na$^+$ in the effluent being applied will penetrate the soil over time and saturate the soil solution with Na$^+$. Displacement of exchangeable cations Ca$^{2+}$ and Mg$^{2+}$ by Na$^+$ will occur in the soil due to the ability of smaller, numerous and more soluble Na$^+$ cations being able to attain an exchange site ahead of the larger, less numerous and less soluble Ca$^{2+}$ and Mg$^{2+}$ cations. With respect to Na$^+$, the dominant cation in effluent, this occurs until equilibrium between the permeant and the soil is reached, leaving the soil relatively higher in Na$^+$ and increasing the ESP. In reality, the soil comes to exchange equilibrium with the concentration of cations in the applied waters.
2.2.3 The Exchangeable Sodium Percentage (ESP) and Sodium Adsorption Ratio (SAR).

Exchangeable sodium is widely determined by soil extraction techniques as outlined in the *Australian Laboratory Handbook of Soil and Water Chemical Methods* (Rayment and Higginson, 1992). The soil exchangeable sodium percentage (ESP) (Equation 1) represents the ratio of exchangeable Na\(^+\) cations to the Cation Exchange Capacity (CEC - the sum of exchangeable Ca\(^{2+}\), Mg\(^{2+}\), Na\(^+\), K\(^+\), Al\(^{3+}\) and H\(^+\) in cmol(+)/kg):

\[
ESP\% = \frac{[Na^+]}{(CEC)} \times 100 \quad \text{(Equation 1)}
\]

The sodium adsorption ratio (SAR) is a solution equivalent to the ESP using the relative molar proportions of Na\(^+\), Ca\(^{2+}\), and Mg\(^{2+}\) (measured in meq/L) in the irrigation waters to be applied (Equation 2):

\[
SAR = \frac{[Na^+]}{\sqrt{(Ca^{2+} + Mg^{2+})/2}} \quad \text{(Equation 2)}
\]

Because of its high solubility in soil solution, potassium (K\(^+\)) is usually neglected as it rarely occupies exchange sites (McBride, 1994). The ESP and SAR are related to the process of cation exchange (Rengasamy and Olsson, 1991; So and Aylmore, 1993; Sumner, 1993). The SAR also describes the probability of Na\(^+\) attaining an exchange site in the receiving soils of a given ESP. The ESP value can be expressed as an SAR equivalent by a number of relationships. These include the equations proposed by (Richards, 1954):

\[
ESP = [100(0.0126 + 0.01475 \times SAR)] ÷ [(1 + (−0.0126 + 0.01475 \times SAR))] \quad \text{(Equation 3)}
\]

and by McBride (1994):
Although there are a number of ESP/SAR relationships available laboratory, it has been shown that ESP and SAR values between 0 and 32 are approximately equal (Quirk, 2001). In Australia, it is commonly accepted that an ESP of 6 % is the lower limit of a sodic soil, while 15 % is considered high (White, 1997). In contrast, the USA has a lower limit of 15 %, which was based on soil permeability experiments in the past which did not take into account the total electrolyte concentration of the potable water used during these experiments, thus soil degradation occurred at a much higher apparent ESP (Sumner, 1993).

So and Aylmore (1993) questioned the validity of ESP threshold limits based on the fact that exclusive use of ESP as a measure of Na\(^+\) in the soil could be misleading over a range of soil types, climate and land-use. Quirk (2001) also disagrees with threshold ESP values being used. The fundamental principles emanating from Quirk and Schofield (1955) state that:

“\textit{Irrespective of the degree of sodium saturation, it is possible to maintain soil permeability by choosing the appropriate electrolyte level in the irrigation water.}”

and also that

“\textit{Since the permeability of a sodium-affected soil is related to the electrolyte concentration of the irrigation water, there is no particular basis for the division of soils into sodic and non-sodic classes at 15 \% exchangeable sodium percentage (ESP).}”

2.2.4 Charged soil particles, micro-aggregate stability and soil water electrolyte concentration

As previously mentioned, high exchangeable Na\(^+\) in soils is strongly associated with poor structural characteristics, which are more pronounced in soil types with higher
clay contents as interstitial pore space is more restrictive and subject to stronger inter-particle forces.

Past research has shown swelling and dispersion of the clay component in soils to be induced by inter-particle forces affecting soil structure, permeability, and other soil properties (Shainberg and Letey, 1984; Rengasamy and Olsson, 1991; So and Aylmore, 1993; Quirk, 2001). In effect, the clays in soils are structured negatively charged particles, so it is important to explain the influence that increased exchangeable Na$^+$ concentration has on charged particle behaviour in soil and soil water.

Verwey and Overbeek (1948) first described the distribution of ions at charged interfaces in aqueous solutions. For negatively charged surfaces (as in a clay particle), positively charged ions (cations) will counterbalance the negative surface charge by experiencing electrical forces of attraction towards the surface. At the same time, Brownian (thermal) motion causes the cations to diffuse away from the surface (Verwey and Overbeek, 1948). The concentration of cations decreases with distance from the negatively charged surface until it becomes equal with the concentration of the surrounding solution in which it is contained.

The surrounding solution is often referred to as the Outside Solution (OS). The diffuse distribution of cations together with the negative surface charge is known as the Gouy-Chapman double layer, named after the two scientists who mathematically described the diffuse space charge. The Gouy-Chapman double layer and associated physics is commonly known as Diffuse Double Layer (DDL) Theory (van Olphen, 1977; Quirk, 1994).

With respect to soils, the negatively charged particles are referred to as clay plates (McBride, 1994). A series of clay plates electrically bound in solution has been referred to as a clay domain or quasi-crystal (White, 1997) and are more pronounced in Ca$^{2+}$-dominated structures (Lunen, 1993). These inevitably reside with other matter in the soil profile. In the soil substrate and accompanying soil water, the clay particles are assumed “compressed” and will exist in natural soils of all textures (Quirk, 2001).

For soils with a texture grade lighter than a clay loam, the interaction between clay particles is accommodated within the interstices between coarser particles. An interpretation of this is given by White (1997) and shown graphically in Figure 2.1.
Figure 2.1: Possible arrangements of clay domains, organic polymers and quartz grains in a micro-aggregate (after White, 1997).

The extent of the thickness of the DDL is governed by the electrolyte concentration of the OS, with its thickness at a maximum when in equilibrium with the OS (McBride, 1994). DDL theory states that the more concentrated the OS, the smaller the distance of the double layer. Conversely, the lower the electrolyte concentration of the OS the larger the double layer within clay structures. Both concepts are represented in Figure 2.2.

The thickness of the double layer is measured with reference to $1/\kappa$, the Debye characteristic length in the Debye-Huckel theory of strong electrolytes (White, 1997). Table 2.1 shows the relative difference in Debye length of the DDL when using Na$^+$ and Ca$^{2+}$ dominated solutions, with the Na$^+$ solution creating larger distances between negatively charged clay plates over a range of concentrations.
Figure 2.2: Inter-crystalline or domain swelling of a Ca-saturated 2:1 clay (after White, 1997)

Table 2.1: Relative difference in Debye length between Na and Ca electrolytes (after Quirk, 2001)

<table>
<thead>
<tr>
<th></th>
<th>0.1M</th>
<th>0.01M</th>
<th>0.001M</th>
</tr>
</thead>
<tbody>
<tr>
<td>NaCl</td>
<td>9.6</td>
<td>30.4</td>
<td>97</td>
</tr>
<tr>
<td>CaCl₂</td>
<td>5.6</td>
<td>17.6</td>
<td>56</td>
</tr>
</tbody>
</table>

Clay domains can be represented by the 3-plate model (Figure 2.3), which illustrates the relationship between individual clay plates within a clay structure. Figure 2.3 shows the conceptual arrangement of clay plates in a clay domain, where one clay plate separates two other clay plates to form a slit-shaped pore. This pore is assumed to be approximately equal to the thickness of the clay plate, although in reality, it will vary
in thickness. The area of plate overlap will also vary so there will be a size distribution of slit-shaped pores (Aylmore and Quirk, 1962, 1967; Murray et al., 1985; Quirk, 2001).

Figure 2.3: 3-plate model of a clay domain (after Quirk, 2001).

The slit shaped pore can be seen to be of the same size as the thickness of the clay crystals. Within these pores classical repulsive DDL forces operate (Quirk, 2001). In the regions of crystal overlap, recently discovered strong attractive forces operate (Quirk, 2001). The balance between these forces (area multiplied by pressure) determines the stability of the clay domain and is easily influenced by cation inputs sourced from irrigation waters (Shainberg and Letey, 1984).

Studies on different clay minerals (Ca/Na systems) using monovalent (Na⁺) and divalent (Ca²⁺) ion solutions have provided insight into the mechanisms responsible for clay stability (Aylmore and Quirk, 1967; Shainberg et al., 1971; Shainberg and Caiserman, 1971; Rengasamy, 1982; Rengasamy et al., 1984; Slade et al., 1991; Chorom and Rensgamy, 1995) and the subsequent impacts on soil hydraulic properties (Quirk and Schofield, 1955; Frenkel et al., 1978; Shainberg and Letey, 1984; So and Aylmore, 1993; Sumner, 1993; Halliwell et al., 2001; Meneer et al., 2001).
As the amount of sodium increases in the soil, the electrophoretic mobility of clay domains increases because of the expanding double layer surrounding the clay domains (Halliwell et al., 2001). However, the clay domains remain intact at ESP values less than approximately 15-25 because of “demixing”, which occurs in Ca/Na systems and results from a non-random distribution of Na\(^+\) and Ca\(^{2+}\) on clay domains (Halliwell et al., 2001). As the ESP rises past 15-25, Na\(^+\) preferentially adsorbs to the outer surface of clay domains and swelling within the domain is predominantly due to hydration, although to a lesser extent, movement of Na\(^+\) into the clay domain (osmotic swelling) occurs from this point (Halliwell et al., 2001).

When the soil ESP exceeds approximately 50, the distribution of both Na\(^+\) and Ca\(^{2+}\) is uniform and swelling (hydration and osmotic) is the same as for pure Na\(^+\) systems (Sumner, 1993). Because of the demixing phenomenon, it is thought that dispersion is the dominant process occurring at low ESP values, whereas swelling is thought to be dominant at high ESP values (Halliwell et al., 2001). Demixing is represented in Figure 2.4.

Figure 2.5 compares Na\(^+\) and Ca\(^{2+}\) dominated clays, showing the difference in stacking arrangements that affect micro-aggregate stability, hence soil pore structure and permeability. Note the slit-shaped pore size is reduced and clay plates are further apart in the Na\(^+\) clay. It is this phenomenon (osmotic swelling) that follows initial hydration swelling, which decreases soil permeability and is due to Ca\(^{2+}\) being displaced by Na\(^+\) in the clay domain. In the case of irrigation with secondary treated effluent, Na\(^+\) will accumulate over time and can become increasingly problematic if not properly monitored (Bond, 1998).
Figure 2.4: Comparison of particle arrangements in a homoionic Na-montmorillonite (right) with that in a Ca-Na system (left) illustrating the formation of clay domains with "demixing" of Na and Ca (after Sumner, 1993).

Figure 2.5: Plate stacking arrangements in Na\textsuperscript{+} clay and Ca\textsuperscript{2+} clay (after McBride, 1994)
Soil swelling, dispersion and deflocculation of clay minerals have been implicated in the assessment of reduced permeabilities. Swelling of Na-smectite may be of such a magnitude as to cause considerable alteration to soil matrix characteristics (porosity, pore size distribution, tortuosity and void space) (Cass and Sumner, 1982).

Due to a decrease in permeant concentration, Quirk and Schofield (1955) proposed that swelling could result in the blocking or partial blocking of conducting pores. By demonstrating a close relationship between the threshold concentration \( C_{TH} \) at which 10-15% reduction in permeability occurred and the commencement of swelling in soil pads, it has also been shown that swelling is the initial cause of reductions in hydraulic conductivity \( K \) (Rowell et al., 1969).

McNeal et al. (1966) found a linear relationship between \( K \) reduction and macroscopic swelling, by relating clay swelling measurements to theoretical interlayer swelling values calculated from data published by Norrish (1954), where correlation coefficients ranged from 0.94 to 0.97 (cited in Cass and Sumner, 1982). McNeal et al. (1966) postulated that \( K \) reductions in sodic soil involved a two-step process, with initial swelling followed by particle dispersion and translocation at lower concentration. In coarse textured soils and those containing 1:1 layer silicates, the dominant mechanism of \( K \) reduction proposed by McNeal et al. (1966) and confirmed by Frenkel et al. (1978) is dispersion and deflocculation. It must be noted that McNeal et al. (1966) did not specifically distinguish between aggregate failure and dispersion and deflocculation (Cass and Sumner, 1982).

Aggregate failure generally occurs at relatively high concentration values as a consequence of heterogeneous distribution of applied external pressure coupled with internal swelling pressures developed during percolation (Cass and Sumner, 1982). This can lead to deformation of aggregates (particle movement) when swelling pressure exceeds yield stress of the aggregates (Cass and Sumner, 1982; Halliwell et al., 2001; Quirk, 2001). Dispersion and deflocculation occur only at a later stage, when swelling has caused sufficient particle separation for attractive forces to be exceeded by repulsive forces, leading to disintegration of aggregates, translocation of dispersed clay particles downward through conducting pore space and subsequent decreases in permeability (Cass and Sumner, 1982).
2.2.5 Soil permeability with respect to the threshold concentration ($C_{TH}$) and turbidity concentration ($C_{TU}$)

Quirk and Schofield established the qualitative relationship between soil ESP and electrolyte concentration with respect to permeability (cited in Quirk, 2001). They showed that laboratory permeability was a function of the relative proportion of exchangeable sodium or of a related solution parameter, the SAR. In their study, Sawyers 1 soil was subjected to electrolyte solutions (both NaCl and CaCl$_2$) with a range of SAR values.

For each SAR value, the soil was brought to equilibrium in a solution that was sufficiently concentrated to sustain a constant permeability. Electrolyte concentration was then reduced in a step-wise fashion while maintaining a constant SAR value. When permeability decreased by 15%, Quirk and Schofield (1955) deemed this the $C_{TH}$, for each SAR value.

Further reductions in the electrolyte concentration of the percolating solution resulted in the appearance of dispersed clay particles (Table 4 in Quirk and Schofield, 1955) and this was designated the $C_{TU}$. Note that the SAR values used in Equations 5 and 6 are converted ESP values from measured soil analyses and are not the actual electrolyte concentrations of the applied water. The $C_{TH}$, with respect to SAR values in the range of 0 - 32, is given by Equation 5 (Quirk, 1971):

$$ (C_{TH}) = 0.56 \times \text{SAR} + 0.6 \quad \text{(SAR 0 - 32)} \quad \text{(Equation 5)} $$

The $C_{TU}$, with respect to SAR values in the range of 0 - 32, is given in Equation 6 (Quirk, 1984):

$$ (C_{TU}) = 0.16 \times \text{SAR} + 0.2 \quad \text{(SAR 0 - 32)} \quad \text{(Equation 6)} $$

Reeve (1958) re-plotted some of Quirk and Schofield’s (1955) results, which are provided in a modified form (from Quirk, 2001) in Figure 2.6. This shows that a series of curves for the same soil can be obtained using permeants of differing electrolyte...
concentrations. It is important to note that as ESP increases, the electrolyte concentration of the applied solution must also increase to maintain optimum permeability.

For example, Davidson and Quirk (1961) demonstrate the impact of changing the electrolyte concentration of irrigation waters, using Riverina clay (60% clay, pH=7.4, ESP=23) near Deniliquin, NSW. The soil was irrigated with waters that had an electrolyte concentration slightly higher than the $C_{TH}$ (point A in Figure 2.6) and with Murrumbidgee River water, which was approximately half the $C_{TU}$ (point B in Figure 2.6). In the first case, the 7.5 cm of water applied was observed to have permeated completely into the soil after 16 hours (Quirk, 2001). In contrast, large volumes of the Murrumbidgee water remained pooled on the surface after a similar time period.

![Figure 2.6: Permeability as a function of electrolyte concentration and ESP (after Davidson and Quirk (1961), modified from Quirk, 2001).](image)

Quirk (2001) states that when the irrigation water electrolyte concentration exceeds the $C_{TH}$, the soil appears granular and dries to a friable state (flocculated). Conversely, when irrigation water electrolyte concentration is less than the $C_{TU}$, the
surface soil appears white (dispersed clay particles) and water remains pooled on the surface for extended periods (Quirk, 2001).

In addition, there appears to be an error in the original figure given in Quirk (2001). The y-axis scale is stated as \(10^4 \times \text{Permeability (cm/sec)}\), and appears unrealistically large for soil permeability of an arid heavy clay. For example, the curve for the soil at ESP = 5.8 indicates that a permeant of electrolyte concentration 4 mmol(+)L will permeate at approximately 20 000 cm/sec! Therefore, it was assumed that the y-axis should have been \(10^{-4} \times \text{Permeability (cm/sec)}\) and this is shown in Figure 2.6.

The magnitudes of the \(C_{TH}\) and the \(C_{TU}\) used in previous studies have been determined for a range of soil textural classes. The \(C_{TH}\) and the \(C_{TU}\) determined for a non-arid loam (Quirk and Schofield, 1955) are equally applicable to the arid heavy clay used in the study by Davidson and Quirk ((1959) cited in Quirk (2001)). Also, Rengasamy et al. (1984) characterized spontaneous dispersion by defining an empirical linear function relating SAR (as \(\text{SAR}_p\)) to the electrical conductivity (EC) required for flocculation. The \(\text{SAR}_p\) is a result of analysing a 1:5 extract (in distilled water) after 1 hour of shaking, then using Equation 2 (Davidson and Quirk, 1961), as opposed to other extraction solutions found in Rayment and Higginson (1992).

Rengasamy et al. (1984) studied a wide range of Australian red-brown earths and obtained an equation, almost identical to Equation 6, to describe the electrolyte concentration at which spontaneous dispersion occurs. They also observed surface turbidity after rainfall (electrolyte concentration \(< C_{TU}\)), even though the soil was protected by pasture.

Therefore, the \(C_{TH}\) and the \(C_{TU}\) are significant and are useful in defining dispersion, hence micro-aggregate stability/soil pore structure and relative permeability, over a large range of soil types. Many guidelines based on the \(C_{TH}\) concept have been suggested to avoid the adverse effects on water transport in sodic soils (Quirk, 1971; Cass and Sumner, 1982; Rhoades, 1982; Jayawardane, 1992; Jayawardane et al., 2001). However, the \(C_{TH}\) concept is not a unique function of SAR and electrical conductivity (EC), but varies with soil type and other factors (Sumner, 1993; Balks et al., 1998).
factor that influences $C_{TH}$ and $C_{TU}$ is the percentage of charged soil particles (clay minerals) present in the soil profile.

Based on the work of Shaw et al. (1994), soils containing 40–50% clay and mixed mineralogies with limited shrink-swell ability, form the most dense soil matrix and have reduced infiltration and plant available water capacity. Soils of this type appear to be the most difficult to manage. As a result, soil behaviour rather than specified ESP level is a better estimate of sodicity effects (Shaw et al., 1994). Soil response to irrigation waters will thus depend on the temporal and spatial presence of solutes.

Mechanisms for cation exchange and displacement will occur within interstitial pore spaces and on the internal/external surfaces of clay minerals. As a result, micro-aggregate/soil pore stability is continually being altered and will impact upon soil permeability over time, particularly when sodium rich waters are consistently applied. The equilibrium of specific cations in the soil solution with a clay surface will be a function of pH and the cations present. Therefore, at one extreme there is a soil profile shown in Figure 2.7, which has a low in pH (<< 4.5) and has been leached of accompanying cations.

![Figure 2.7: A highly acidic soil with leached cations (Ca$^{2+}$, Mg$^{2+}$, Na$^+$, K$^+$) and associated clay particle response](image)

The removal of cations by decreasing pH represents an acidic soil, with Ca, Mg and Na displaced by Al$^{3+}$ and H$. Al$ is drawn from within the clay particle. As a result of this cation imbalance, soil structure is relatively stable however is agriculturally unproductive due to potential Al toxicity.

Figure 2.7: A highly acidic soil with leached cations (Ca$^{2+}$, Mg$^{2+}$, Na$^+$, K$^+$) and associated clay particle response
When monovalent and divalent cations occupy clay surfaces, the soil profile may be favourable for plant production. As a result, micro-aggregate/soil pore stability is again altered based on DDL theory however remains stable due to the increased relative presence of Ca$^{2+}$ over Mg$^{2+}$ and Na$^+$ (Figure 2.8).

![Soil Water Condition and Soil Structure Response Diagram](image)

The balance of cations represent a productive soil, with Ca being dominant and Mg and Na in progressively lower concentrations respectively. Al is bound within the clay matrix and will only be released with a major decrease in soil pH (< 4.5). Ca, Mg and Na are constantly competing for soil exchange sites and as soil pH values > 5, these can be conceptualised as the dominant cations potentially involved in altering soil pore structure and optimum permeability. As a result of this cation balance, soil structure is stable and defines optimum transmission of water through the soil profile.

Figure 2.8: An agriculturally “productive” soil with a balance of cations and associated soil response

When sodium accumulates and displaces Ca$^{2+}$ and Mg$^{2+}$, further alteration to micro-aggregate/soil pore occurs, resulting in a further decrease from optimum permeability. This is predominantly due to the decrease in diameter of soil pores as clay particles expand and reduce effective transmission of water downward through the soil profile.
However, the type of clay mineralogy has been shown to influence $\text{Na}^{+}$-$\text{Ca}^{2+}$ selectivity. The influence of clay mineralogy plays a major role in $\text{Na}^{+}$-$\text{Ca}^{2+}$ selectivity, that is, the preferential attraction of monovalent or divalent cations to a given clay surface. The study of Kopittke et al. (2005) has shown that $\text{Na}^{+}$-$\text{Ca}^{2+}$ exchange varies with both ionic strength and clay mineralogy. In their study, the changes were explained through the application of DDL theory in clay exchange surfaces. Clays with predominantly external exchange sites (e.g. kaolinite and pyrophyllite) showed preference for $\text{Na}^{+}$ by compressing the DDL and decreasing electric potential. The magnitude of this preference increased with decreasing ESP (Kopittke et al., 2005).

Conclusions by Kopittke et al. (2005) state that soil ESP approximations determined from the generalised SAR-ESP equation (Richards, 1954) are not constant and actually vary with ionic strength and clay mineralogy. Therefore, investigating the extent of sodium dynamics (SAR of applied waters and soil ESP) will provide greater insight into the underlying continuum that governs temporal soil structure in an effluent irrigated woodlot soil.

![Figure 2.9: A sodium-dominated soil profile and subsequent response of clay particles](image)
2.2.6 Solute transport in soil water

Two main processes affect the transport of solutes in soil water, namely advection and hydrodynamic dispersion (Hudak, 1999). Advection occurs when flowing water transports solutes. Hydrodynamic dispersion is the spreading of soil water and solutes by mechanical mixing and diffusion (Hudak, 1999). Mechanical mixing in soil water is caused by deformation of the soil matrix; for example, the change in soil pore structure caused by changes in the SAR of the soil water (Levey, 1984).

Diffusion is where solutes flow from a region of high concentration to a region of lower concentration until equilibrium is reached (Hudak, 1999). For example, a drop of coloured dye placed into a beaker of clear, stagnant water will spread evenly throughout the solution over time. The continuity condition for one-dimensional miscible transport through a soil column, including adsorption but excluding other physical/chemical reactions such as radioactive decay and precipitation and/or biological reactions such as bio-degradation, may be written as follows (Shackelford, 1994):

\[
\frac{\partial (\theta R_d c_r)}{\partial t} = -\frac{\partial J}{\partial x}
\]  
(Equation 7)

where \( \partial = \text{“change in”}, \theta = \text{volumetric soil water content}; c_r = \text{solute concentration in the pore water of the soil}; R_d = \text{retardation factor accounting for linear, instantaneous and reversible equilibrium sorption of reactive solutes}; t = \text{time}; x = \text{macroscopic distance in the direction of transport}; \) and \( J = \text{the solute flux in the soil} \) (Shackelford, 1994). The concentration, \( c_r \), has been defined as a resident concentration or a volume-averaged concentration representing the mass per solute per unit volume of fluid contained in an elementary volume of the soil at a given instant (Shackelford, 1994).

In the absence of coupled flow processes the solute flux \( J \), is given by (Shackelford, 1994):

\[
J_a = J_d (x,t) = \theta \left[ v c_r (x,t) - D \frac{\partial c_r (x,t)}{\partial x} \right]
\]  
(Equation 8)
where $D =$ hydrodynamic dispersion co-efficient and $v =$ seepage velocity equal to $q/\theta$
where $q$ is the fluid flux given by Darcy’s law. The first term represents the advective
solute flux, $J_a (= \theta v c)$ and the second term represents the dispersive solute flux $J_d (= - \theta \frac{D c_r}{\partial x}$), including the flux due to diffusion. For saturated soils in which all of the void
space is effective in conducting fluid flow, $\theta$ equals the total porosity (Shackelford, 1994).

If $v, \theta, R_d$ and $D$ are assumed constant, then the combination of Equations 7 and
8 results in the well known advection-dispersion equation describing one-dimensional
transient solute transport through homogenous soil, or:

$$R_d \frac{\partial c_r}{\partial t} = D \left( \frac{\partial^2 c_r}{\partial x^2} \right) - v \left( \frac{\partial c_r}{\partial x} \right) \quad \text{(Equation 9)}$$

Concentration-based analytical solutions to equation 9 in the form $c_r = c_r(x,t)$
have been investigated and derived for several different initial and boundary conditions
(van Genuchten, 1981; van Genuchten and Alves, 1982). Equation 8 shows that sodium
flux can be related to volumetric moisture conditions ($\theta$), particularly under saturated
conditions. Subsequently, it can be assumed that soil permeability would vary under
irrigation waters of varying SAR, affecting water movement through the soil profile.
This has ramifications for woodlot soil structure over time, particularly when irrigated
with secondary treated effluent.

2.2.7 Implications of changing soil ESP / effluent SAR for micro-aggregate stability.

When the electrolyte concentration (as a SAR) of the applied water increases, the
charge in the soil’s OS also increases (reducing particle-particle repulsion) and clay
plates begin to overlap. Physically, the aggregation of clay plates occurs because of a
lowering of the free energy of the soil solution, causing the inter-layer spacings between
clay particles to decrease (Shainberg et al., 1971; Shainberg and Caiserman, 1971;
Loveday, 1976; Rengasamy, 1982).
A further increase in electrolyte solution will cause groups of overlapped clay plates to agglomerate and fall out of solution. The agglomeration of charged particles is called flocculation, and can be caused by both high-charge cations and high-electrolyte (salt) solutions (Marwan and Rowell, 1995). Several flocculated forms of clay plates have been identified, including face-to-face, edge-to-face, and edge-to-edge arrangements that appear to be governed by clay domain size and the pH of the soil water (White, 1997). In a flocculated state ($\geq C_{TH}$), the arrangement of the clay plates (illustrated in Figure 2.10) causes a decrease in micro-porosity, which consequently causes lower permeability when compared to the pre-irrigation permeability.

Dispersed clay plates occur when the outside solution concentration is much less (approximately equal to $C_{TU}$), resulting in clay plates blocking pore space between micro-aggregates immediately below the dispersion layer and significantly decreasing permeability. Fireman (1944) found that permeability dropped to approximately 1% of its initial value when a Hisperia sandy loam (ESP = 20) was eluted with distilled water after previously being eluted with 800 ppm CaCl$_2$ (Quirk, 2001). However, Fireman (1944) did not recognise the significance of this at the time (cited in Quirk, 2001).

Figure 2.10: Edge-to-face flocculation of kaolinite at low pH (after White, 1997)
Several conditions are known to assist in maintaining micro-aggregate stability. These include a decrease in pH of the soil water (Suarez et al., 1984), an increase in sesquioxides (ie, Al$^{3+}$, Fe$^{3+}$) and/or an increase in organic matter (Churchman et al., 1993; Nelson et al., 1999; Lado et al., 2004). Interestingly, each of the above represents a case of electrolyte concentration maintaining soil structure.

A drop in pH means an influx of H$^+$ ions, whose exchange energy is sufficient to remove all cations from soil exchange sites, exposing the negative faces of clay particles. Since all electrolytes are now in solution, they impart a stronger cationic charge energy that causes flocculation (McBride, 1994; White, 1997). Similarly, the presence of high charge sesquioxides of aluminium (Al$^{3+}$) and iron (Fe$^{3+}$), are sufficient to buffer against mild sodium accumulation effects (McBride, 1994).

Organic matter assists micro-aggregate stability by using humic/fulvic acids that stabilise pH and by providing an environment that stimulates bioturbation, aeration and permeability (Levy et al., 1999; Nelson et al., 1999). Also, colloidal organic matter has been shown to block soil pore continuity (Magesan et al., 1999).

Micro-aggregate/soil pore stability has been classified by numerous methods in the past (Emerson, 1967; Cass and Sumner, 1982; Rengasamy et al., 1984; Emerson, 1991). Emerson (1967) developed the Emerson Aggregate Test (EAT), which involves immersing aggregates (approximately 2 to 5 mm in diameter) in distilled water. The EAT has 8 classifications based on slaking, swelling, mechanical disturbance, carbonates, flocculation and dispersion. An Emerson Class 1 aggregate is highly unstable and can be described as dispersive, whereas an Emerson Class 8 aggregate does not slake or swell and can be described as stable.

Emerson has since reviewed the criteria of each class, with respect to aggregate stability during agricultural activities, eliminating the Class 8 classification (Emerson, 1991). The issue with using the EAT as an assessment tool for effluent irrigated sites is that the SAR of irrigation waters is much higher than distilled water. The standard EAT test uses distilled water as an immersion solution and therefore is limited in its application to effluent irrigated sites.
More recently, an amended EAT using an SAR5 solution has been used to better assess the micro-aggregate stability of soils planned to receive STE (Patterson, 2001). In the above tests, the measured degree cannot be related unambiguously to ESP or electrolyte effects (Rengasamy et al., 1984). However, in the management of dispersive soils, it may be necessary to distinguish between the two (Rengasamy et al., 1984). Overall, it appears that the EAT would yield different results for immersion solutions of differing SAR values, although it appears useful when the SAR of the irrigation waters is known and tested accordingly. Understanding the intrinsic relationship between both electrolyte concentration and the SAR of the irrigation waters is important for assessing soil structure and will be discussed in detail in Section 2.2.8.

Although not specifically to address this “continuum” of classification, Cass and Sumner (1982) modelled soil pore structural stability based on irrigation waters of differing SAR values. To do this, they determined the relationship between the slopes of the threshold concentration ($C_{TH}$) curve and associated permeability reductions. This defined a less arbitrary means of evaluating micro-aggregate stability, which Cass and Sumner (1982) called the sodium stability curve.

The slope of the sodium stability curve reflects the resistance of the soil to reductions in $K$ while the intercept indicates the sensitivity of soil pore structure to changes in solution concentration (Cass and Sumner, 1982). The same concept can be applied to the relationship between solution composition and macroscopic swelling of soils or extracted clays (Cass and Sumner, 1982).

Jayawardane (1992) predicted the unsaturated hydraulic conductivity of a loamy soil using the equivalent salt solution series. The equivalent salt solution series is based on the findings of Quirk and Schofield (1955) and McNeal and Coleman (1966), where the extent of swelling and saturated hydraulic conductivity values of a soil were similar in high salt solutions at all SAR values investigated. The equivalent salt solution series relates equivalent changes in soil pore geometry to an equivalent salt solution of specific combination of SAR and EC. As a result, changes in unsaturated hydraulic conductivity can be predicted for a soil irrigated with a known salt solution.
Although useful for determining optimum permeability, the sodium stability curve and equivalent salt solution do not incorporate the misconception highlighted by Quirk (2001). That is, unsaturated hydraulic conductivity (in studies by Jayawardane (1992) and others) does not take into account the decrease in permeability due to solutions with SAR values less than $C_{TU}$. Also, irrigation waters are rarely applied in a way as to sustainably increase salt concentrations in maintaining optimum permeability.

In researching soil physical/chemical properties and the behaviour of the clay components under different electrolyte concentrations, it became evident that a continuum existed between soil ESP and the SAR of the applied irrigation waters. Crescimanno et al. (1995) have also suggested that a continuum may exist between soil structural properties and soil ESP. One study has shown that an ESP as small as 2 to 5 can cause adverse effects if low sodium concentrations are present in the soil solution (Meneer et al., 2001).

2.2.8 Electrolyte concentration and SAR – understanding the relationship for managing effluent irrigated woodlots.

The use of either electrolyte concentration or SAR as an indicator for predicting the response of irrigated soils must be differentiated. For effluent irrigated woodlots, the application waters rarely consist of just one electrolyte (ions in solution). Application waters usually consist of many electrolytes, the most dominant cations being sodium, calcium, magnesium and potassium. In this thesis, the sum of these cations (in meq/L) constitutes the electrolyte concentration. Thus, the response of soil structure can be predicted using electrolyte concentration and this has been validated by the work of Quirk and Schofield (1955).

The relationship between electrolyte concentration and SAR can be more complex. For example, it is possible for a solution to increase and/or decrease in electrolyte concentration, whilst maintaining a constant SAR. It is also possible for a solution to increase in SAR and increase in electrolyte concentration. However, it appears highly unlikely that a solution with an increasing SAR would undergo a decrease in electrolyte concentration in the natural environment.
The most common process resulting in a decrease in SAR and electrolyte concentration is dilution by rainfall, which causes a uniform decrease in individual electrolytes. Rainfall appears to be the most damaging to soil structure, as both the electrolyte concentration and SAR of rainfall are much less than that of the applied effluent.

Therefore, if variations in SAR can be related to variations in electrolyte concentration, the use of the SAR to determine the response of woodlot soils to effluent irrigation appears sound. In addition, the SAR better describes the proportion of electrolytes in a solution, as sodium is the main cation to be managed in the effluent irrigated woodlot studied.

2.3 Effluent irrigation of woodlots

2.3.1 Effluent characteristics

This study deals with low strength Secondary Treated Effluent (STE) (EPA., 1995) originating from municipal wastewater. Table 2.2 shows the typical chemical characteristics of secondary treated effluent (data sourced by Myers et al. (1999) from New South Wales Land and Water Conservation (1997) and Falkiner and Smith (1997)).

<table>
<thead>
<tr>
<th>PARAMETER</th>
<th>MEAN</th>
<th>RANGE</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>8.2</td>
<td>6.8 – 9.6</td>
</tr>
<tr>
<td>EC (dS/m)</td>
<td>0.8</td>
<td>0.46 – 1.5</td>
</tr>
<tr>
<td>Total N (mg/L)</td>
<td>11.2</td>
<td>2 – 50</td>
</tr>
<tr>
<td>Total P (mg/L)</td>
<td>6</td>
<td>2 – 18</td>
</tr>
<tr>
<td>Ca(^{2+}) (mg/L)</td>
<td>26</td>
<td>11 – 55</td>
</tr>
<tr>
<td>Mg(^{2+}) (mg/L)</td>
<td>19</td>
<td>5 – 50</td>
</tr>
<tr>
<td>Na(^+) (mg/L)</td>
<td>132</td>
<td>50 – 250</td>
</tr>
<tr>
<td>Chloride (mg/L)</td>
<td>124</td>
<td>40 – 350</td>
</tr>
<tr>
<td>Sodium Adsorption Ratio (SAR)</td>
<td>4.8</td>
<td>3.8 – 7.1</td>
</tr>
</tbody>
</table>

Municipal wastewater usually has a low to medium salinity (EC of 0.6 – 1.7 dS/m), accompanied by a high concentration of sodium relative to the other cations.
(Ca$^{2+}$, Mg$^{2+}$ and K$^+$) (Myers et al., 1999). While sodium concentrations remain higher than other cations, they are likely to cause an increase in the soil exchangeable sodium percentage (ESP), increasing the potential of soil deterioration through swelling and subsequent clay dispersion (Myers et al., 1999). The trigger for soil deterioration has been attributed to a decrease in the electrolyte concentration of the water permeating the soil and/or an increase in the ESP of the receiving soil (Richards, 1954; Quirk and Schofield, 1955; Rengasamy et al., 1984; Sumner, 1993; Halliwell et al., 2001; Quirk, 2001). Furthermore, the response of the receiving soil to variations in the SAR of the irrigation waters will also vary over time.

2.3.2 Effluent irrigation principles

One land disposal option currently being used in Australia and overseas is the effluent irrigation of woodlots (EPA, 1995; Myers et al., 1999). Any large-scale reuse scheme, however, such as irrigated woodlots, must be designed to be sustainable and have no significant impact on the environment. Therefore it is important to initiate monitoring programs for key indicators of soil and groundwater chemistry in order to determine if the practice of effluent irrigation is sustainable over the longer term (Menzies et al., 1999; Myers et al., 1999).

Effluent reuse trials utilising hardwood plantations have been conducted at Bolivar in South Australia (Boardman et al., 1996), Wagga Wagga (Myers et al., 1999) in NSW, Cleveland in south-east Queensland (Moss et al., 1998), Shepparton in Victoria (Duncan et al., 1998) and Rotorua, New Zealand (Tomer et al., 1997). In addition to water balance monitoring and irrigation scheduling work, the movement and fate of nutrients (nitrogen and phosphorus) and, to a lesser extent sodium, in the effluent have been monitored and modelled under the site-specific conditions of each woodlot (Polglase et al., 1995; Falkiner and Polglase, 1997; Falkiner and Smith, 1997; Balks et al., 1998; Myers et al., 1999).

One aspect of the research that has not received significant attention is sodium flux in reuse plantations. With respect to sodium flux, very few woodlot trials have been reported in the literature, although much is available concerning the impact that
Saline irrigation waters have on soil structure. Recent international examples include those by Qadir et al. (2000), Oster and Shainberg (2001) and Shainberg et al. (2001), who comment on saline soil amelioration, soil responses to sodicity/salinity, and the effect of pre-wetting rate and sodicity on permeability, respectively. None specifically refer to effluent irrigated woodlots.

Falkiner and Smith (1997) showed that for a soil growing Pinus radiata and Eucalyptus grandis and irrigated with secondary treated effluent, soil salinity and sodicity increased after four irrigation seasons. Treatments included a high, medium, low salinity effluents and a bore water of similar EC and SAR to the mean of the effluents applied. Soil sodicity increased in response to the addition of effluent and bore water and ESP levels were about 20 – 25 % in the 0.1 – 0.6 m zone (Falkiner and Smith, 1997). Lower ESP values were observed in soils irrigated with lower SAR waters, although the leaching rate was the least of the treatments applied.

Balks et al. (1998) report on increasing ESP in a woodlot soil irrigated with secondary treated effluent. Soil ESP at the Wagga Wagga Effluent Plantation Project was found to increase from less than 2 to greater than 25 at some depths within the surface 0.6 m of soil, with either treated sewage effluent or bore water with similar salinity and SAR. These increases in soil ESP were obtained from only two sampling occasions, over five irrigation seasons, and cannot be assumed to be rising uniformly or otherwise.

The approach by Balks et al. (1998) was based on the need to prevent deep drainage to groundwater of nitrogen and phosphorus rather than remove sodium from the root zone; therefore sodium would be expected to accumulate, as was the case. The amount of sodium leached was estimated from water balance data using the model WATSKED (Myers et al., 1999) and never intensively validated by soil sampling and analysis.

Modelling approaches taken in planning and monitoring effluent irrigation projects have included WATSKED (Myers et al., 1999), MEDLI (Gardner et al., 1997), APSIM-WASTE (McCown et al., 1996) and UNSATCHEM (Simunek and Suarez, 1997), with many simpler input/output spreadsheet versions being utilised by wastewater managers and private users.
WATSKED is an irrigation scheduling model using measured or estimated site-specific parameters such as effluent loading rates, rainfall, pan evaporation, irrigation method (drip, sprinkler or flood), basic soil characteristics, and soil moisture status of the upper and lower root zone. Water balance models, such as WATSKED have relied on a leaching requirement to remove salt from the soil profile (Myers et al., 1999), however the ratio of cations has been shown to influence soil pore stability, hence relative permeability, as discussed in the previous section.

The leaching requirement has been determined from water balance calculations and not so much on the actual flux of sodium within the soil profile, or with little concern for variations in soil permeability that variations in the sodium flux may trigger. Leaching requirements are based on assumptions of steady-state and of absolutely uniform conditions of irrigation, infiltration, leaching and evapotranspiration, although most of these are not achieved under field conditions (Qadir et al., 2000).

2.3.3 Hydraulic and Nutrient Loading

Figure 2.11 shows the main components of the water balance that must be either measured or estimated in order to determine irrigation requirements (Myers et al., 1999).

![Figure 2.11: The main components of the water balance in an effluent irrigated woodlot (after Myers et al., 1999).](image)
In order to not disrupt the water balance in an effluent irrigated woodlot, it is necessary that the reuse scheme be designed to assimilate the applied water (and nutrients). Over-irrigation can induce surface runoff and/or waterlogging while under-irrigation causes stress to the trees. Therefore, the effluent application rate must be closely related to the water used by the trees on a temporal basis (Potential evapotranspiration (PET), whilst also maintaining sufficient soil water in the root zone to promote sodium leaching.

The basic hydraulic criteria regulated by the NSW EPA (1995) regarding effluent reuse, are that surface runoff should be avoided and minimal deep drainage should occur. Therefore, in the design of effluent to land application schemes, a water balance calculation is required based on the following equation:

\[ \text{Applied Effluent} + \text{Precipitation} \leq \text{Evapotranspiration} + \text{Percolation} + \text{Runoff} \]

\[ (Q_e) \leq (E + T) \]

(Equation 10)

The irrigation deficit/surplus (\( \Delta I \)) is determined by re-arranging Equation 10 to equal zero (Equation 11). Zero irrigation deficit/surplus is arbitrarily chosen at a time when the site is deemed saturated (by either secondary treated effluent and/or rainfall), giving a datum irrigation deficit/surplus value of zero. For example, if the datum irrigation deficit/surplus at saturation was equivalent to a volumetric soil moisture of 40%, then a volumetric soil moisture of 20% measured at a later date will provide an indication of relative irrigation deficit/surplus.

The irrigation deficit/surplus is the limiting component for daily irrigation scheduling, which is usually based on the measured volumetric soil moisture at the time of irrigation. A value for interception loss (IL), that is, the percentage of the total hydraulic load not reaching the soil, must also be estimated and included in water balance calculations. As a result, a common water balance used to determine the irrigation surplus/deficit (based on Myers et al. (1999)) is:

\[ \Delta I = [\text{PET} + R + D - ((Q_p + Q_e) - IL)] \]

(Equation 11)
where $\Delta I =$ soil moisture deficit/surplus, $\text{PET} =$ Potential evapotranspiration ($\text{PET} =$ evapotranspiration (ET)), $R =$ runoff, $D =$ deep drainage, $Q_p =$ precipitation, $Q_e =$ applied effluent and $\text{IL} =$ interception loss. All parameters are in millimetres (mm).

A positive value for irrigation deficit/surplus reflects any surplus soil water and any negative value reflects the irrigation deficit. Over time, the cumulative irrigation surplus/deficit defines soil water storage decline and/or recharge. The cumulative irrigation surplus/deficit is a function of the daily irrigation surplus/deficit. For example, if 15 mm of water is lost from the cumulative irrigation surplus/deficit on a given day, then the daily $\Delta I$ would be -15 mm; therefore the cumulative irrigation surplus/deficit = -15 mm. If no effluent/rainfall was applied the following day, and another 15 mm was again lost from the cumulative irrigation surplus/deficit, then the daily $\Delta I$ would still be -15 mm, although the cumulative irrigation surplus/deficit would equal -30 mm.

To determine PET, E is multiplied by a pan coefficient ($K_p$). The $K_p$ is a factor that represents transpiration as a function of pan evaporation. The pan coefficient ($K_p$) for tree water use is assumed to be greater than 1, that is, transpiration is assumed to be greater than evaporation. Trees have higher water uptake potential (larger root zone and leaf surface area) than pasture, for which pan coefficients have been noted as being less than 1 (Brown et al., 2001). The pan co-efficient has been shown to vary throughout the seasons (Honeysett et al., 1992; Myers et al., 1999).

Ideally, effluent irrigation schemes should mimic tree water use and include appropriate assumptions and/or measurements of soil hydrology that promote optimum growth conditions. Optimum growth conditions prevail when the volume of applied water (and nutrients) is utilised by tree demand and when it is in sufficient volume to promote sodium leaching from the root zone (Feign et al., 1991; Nutter et al., 1996; Smith et al., 1996).

A common management practice in effluent irrigation scheduling is to include a leaching requirement to remove excess sodium from the root zone (Qadir et al., 2000). However, implementing a leaching requirement needs some knowledge of the permeability characteristics of any soil receiving irrigated effluent and/or rainfall. Figure 2.12 represents a theoretical soil water management regime (Myers et al., 1999),
highlighting the maximum/minimum volumetric soil moisture used to determine irrigation scheduling.

![Diagram of soil water management](image)

**Figure 2.12: Theoretical soil water management regime in an effluent irrigated woodlot (after Myers *et al.* 1999).**

Conceptually, any given irrigation or rainfall event will have been designed to refill the soil water storage component of the water balance to an appropriate level (usually 90% of field capacity (Myers *et al.*, 1996)), although in practice this is difficult because estimated components of the water balance may cause an inaccurate irrigation schedule (Bond, 1998). As a result, the receiving soil may be under and/or over irrigated with effluent over the short-term.

In theory, sodium will only be leached from the root zone when the solum is saturated (Balks *et al.*, 1998; Snow *et al.*, 1999; Theiveynathan *et al.*, 2004), although this is far from an optimum soil moisture condition for water uptake and tree growth (Battalglia and Sands, 1997). If the solum is only partially saturated, then sodium will only be leached to the depth of effluent and/or rainfall occurring, although this would usually be favourable for tree growth (wilting point $< \theta < $ field capacity) (Honeysett *et al*.)
The limiting rate for irrigation scheduling is two-fold and depends on the depth of effluent and/or rainfall applied and the permeability of the soil.

If the permeability of the receiving soil is less than the rate at which the effluent and/or rainfall occurs, then surface runoff is probable over the longer term. In view of variations in permeability caused by increasing soil ESP, it follows that effluent of varying sodium concentrations applied to soils of fluctuating ESP will also result in deviations from optimum permeability over time. This should be borne in mind when managing and maintaining soil permeability during any effluent irrigation event.

Over a period of successive irrigation cycles, the fluctuation of sodium concentrations with depth has consequences for the stability of micro-aggregates. As a result, permeability decreases and there is a subsequent need to reduce effluent loading rates. In order to prevent surface runoff and promote tree growth, the maximum volume of effluent/rainfall that can be applied at any given time is equal to the volume of water that the soil can hold at field capacity. For example, if the field capacity of a receiving soil is 40 mm, then only 40 mm of effluent/rainfall can be applied. When the soil is near field capacity, the volume of effluent/rainfall must not exceed the volume required to recharge to field capacity, otherwise surface runoff is probable (Shaw, 1994; Hudak, 1999).

Field capacity and wilting point are conceptually shown in Figure 2.12. Field capacity represents volumetric soil moisture at the upper drained limit, whereas wilting point represents the volumetric soil moisture at which plants lose turgor and wilt (White, 1997). When related to a soil moisture curve, both field capacity and wilting point can be implied from direct volumetric soil moisture measurement. These water contents have been found to correspond to soil matrix potentials of between 0.1 and 0.2 bar for field capacity and 10 to 15 bars for wilting point, dependent on soil texture.

Figure 2.13 shows a hypothetical concept of the probable soil sodium flux in relation to applied effluent and/or rainfall permeating to different depths. “SWS” represents the soil water storage component of the site water balance and “Events” A, B, X and Y mm represent irrigation and/or rainfall events of increasing volume. An “Event” can be effluent irrigation, rainfall or a combination of both. The lower boundary of the root zone is also shown. Any event that does not permeate below the root zone...
(Events A and B mm in Figure 2.13) will only allow sodium to move downward equal to the extent of the wetting front produced from that event. Depending on the volumetric soil water content \( (\theta) \), advection and/or hydrodynamic dispersion processes occur. Advection occurs when soil water flow physically transport solutes, whereas hydrodynamic dispersion occurs under low volumetric soil moisture conditions and mainly in interstitial pore space (Freeze and Cherry, 1979; Hudak, 1999).

![Figure 2.13: Probable sodium movement in an effluent irrigated woodlot.](image)

However, trees will uptake availaboratoryle soil water daily and further increase sodium concentrations in the root zone if irrigation scheduling does not satisfy the trees’ requirements. As a result, under-irrigation will lead to sodium accumulation in the root zone. Conversely, any event that permeates below the root zone (Events X and Y mm in Figure 2.13) will inevitably transport excess sodium from the root zone further downward where it may again accumulate (event X mm). When the soil profile is saturated to the water table (Event Y mm), sodium will move into the water table, finally leaching excess sodium from the soil profile. The non-steady state conditions prevailing in most irrigated soils pose difficulties in determining nutrient accumulation/leaching rates.
The aim of sampling/monitoring soil properties over time is to provide information about rates of change as a result of effluent irrigation. The highly variable nature of soil properties makes this a complex task, so a sampling strategy based on soil sampling principles should be established (Myers et al., 1999). The sampling/monitoring of soil physical/chemical properties in effluent irrigated woodlots is usually less frequent than the sampling/monitoring of water balance parameters, due to the fact that nutrients are assumed to accumulate over longer periods.

Most studies illustrate how soils are usually sampled annually, or over longer periods (Stewart et al., 1990; Nutter et al., 1996; Falkiner and Smith, 1997; Hulugalle and Finlay, 2003), making it impossible to assess any temporal flux. For example, if only two samples are taken during a one-year period and wastewater managers want to assess the accumulation/leaching of sodium (as soil ESP), then the outcome would be Figure 2.14. Figure 2.14 indicates that sodium has undergone a net increase over the one-year period.

![Figure 2.14: Example of a plotted annual soil sampling/monitoring assessment.](image)

By increasing the frequency of sampling/monitoring, it is possible to gain greater resolution of sodium flux over the same one-year period (Figure 2.15). For example, it can be seen that there is greater variation in the sodium flux in Figure 2.14, showing sodium is more dynamic than Figure 2.14 reveals.
Both figures used are examples only and have been used to illustrate that by increasing the monitoring resolution of at-a-station temporal soil ESP values, a better understanding of the sodium flux can be gained for a given site. Knowing the range of sodium flux allows for improved discretion in soil management with respect to maintaining optimum permeability and long-term soil structure.

2.4 Summary

This chapter introduced several concepts and theories salient to this study. The significance of the $C_{TU}$, the critical electrolyte concentration defining dispersion of clay particles in micro-aggregates (Quirk, 2001), will be further examined in view of effluent irrigation scheduling.

Quirk (2001) strongly suggests that further soil research should incorporate the $C_{TU}$, as the $C_{TU}$ describes the critical electrolyte concentration and/or soil ESP value for clay particle dispersion, the most damaging process to soil structure. The $C_{TU}$ shows that a degree of flocculated charged clay particles are present under this soil water condition. Therefore, micro-aggregate stability will probably be maintained under the applied condition, that is, an optimum volumetric soil water content and permeant SAR. At the $C_{TU}$, dispersed clay particles will block soil pores and significantly reduce soil permeability. Therefore, it is desirable to minimise and/or prevent “dispersive”
conditions using the range of soil responses provided by the $C_{TH}$ and $C_{TU}$ over time in order to manage soil ESP and maintain optimum permeability.

Optimum permeability of an irrigated soil is characterised by a stable soil structure. A stable soil structure is characterised by the relative composition of various cations within clay particles, namely $Ca^{2+}$, $Mg^{2+}$, $Na^+$ and $K^+$. Different clay mineralogies show a preference for different cations under different conditions (Kopittke et al., 2005). For example, montmorillonite and illite clays show a preference for monovalent ions ($Na^+$) with decreasing electric potential and a preference for divalent ions ($Ca^{2+}$) with increasing potential (Eriksson, 1952; Shainberg et al., 1980).

$Ca – Na$ selectivity is also influenced by the type of exchange surface, which is either an external surface (1:1 phyllosilicate minerals, e.g. kaolinite) or non-expanding 2:1 phyllosilicate minerals (e.g. illite and pyrophyllite) or an internal surface between clay platelets of a domain (expanding 2:1 phyllosilicate minerals e.g. montmorillonite). This suggests that potential cation exchange and/or cation displacement in an irrigated soil are susceptible to the composition of cations in the applied waters, and also where specific clay mineralogies show preference for specific cations. In particular, excess sodium affects micro-aggregate/soil pore stability, which subsequently determines optimum permeability. Therefore, careful interpretation of the amount of rainfall in conjunction with the relative cation composition (as a SAR) and depth of the applied effluent applied to a woodlot soil of specific ESP, will provide insight into a predictive approach to managing optimum permeability in an effluent irrigated soil.

This thesis builds on the irrigation scheduling approach by including a continuum for soil ESP/effluent SAR micro-aggregate/soil pore stability that predicts optimum permeability conditions. The continuum is a result of determining temporal sodium flux in woodlot soils receiving secondary treated effluent and the investigation of the mechanisms responsible for micro-aggregate stability impinging on relative permeability with respect to electrolyte concentration (as a SAR) and soil ESP over time. Quirk (2001) highlights a misconception in the literature, which has been perpetuated since 1955:
“The misconception has arisen that the concentration required to maintain a stable permeability is that which is required to flocculate a dispersed suspension of soil (Felhendler et al., 1974; Frenkel et al., 1978). This perception does not recognise the crucial significance of the turbidity concentration as defined and measured by Quirk and Schofield (1955) and later by Rowell et al. (1969). The turbidity concentration is many times smaller than that required to flocculate a dispersed suspension of soil. The energy imparted to the soil in the preparation of a suspension of dispersed particles destroys the microstructure which is investigated in permeability studies.”

The principle of the nexus between soil ESP and electrolyte concentration (as a SAR) is modelled as a continuum, which has the capability of predicting “dispersive” irrigation/rainfall events over the long term, if the site is sufficiently monitored with respect to sampling frequency and number of sites.

Concepts and theories for soil micro-aggregate/soil pore stability have been reviewed and subsequent decreases in permeability have been related to soil ESP and the SAR of the application waters. Comparison of monitored field results for irrigated and non-irrigated sites will provide the insight into whether short-term variation in soil ESP (flux), impacts upon optimum permeability when irrigated with waters of varying SAR.
Chapter 3: SITE DESCRIPTION AND EXISTING DATA

3.1 Introduction

This chapter describes the selected study site and broadly discusses existing effluent and groundwater data and limited soil analyses as reported by the managers of the site, the Hunter Water Corporation (HWC). In 2002/2003, the Branxton WWTW serviced an equivalent population of 4920 and received approximately 459 ML/yr of municipal effluent (293 L/person/day). Dry weather flow was 1.2 ML/day for the same period, based on pump capacities and hours run. Primary and secondary treatment occurs on-site, in a process of screening, aeration and clarification before discharge to storage ponds. Ponded secondary treated effluent is then irrigated to an adjacent property, Branxton Golf Course or the HWC woodlots. Any surplus effluent is discharged to Anvil Creek.

3.2 Location & topography

The Branxton WWTW is located in the Hunter Valley; approximately 60 km west of Newcastle, NSW (32°21’S, 151°21’E) (Figure 3.1). The study site is bordered by Anvil Creek to the south and south-west and Red House Creek to the east and south-east (Figure 3.2). The New England Highway occupies the higher ground to the north, directing runoff through the site to the southeast (Anvil Creek) or via the swamp to the west bordering the Branxton Golf Course. Elevation ranges from 35 to 40 metres AHD (Australian Height Datum) over the entire site, with a conspicuous drainage line occurring in a narrow depression through Area 3 (Figure 3.2).

The main limitations identified during initial assessment of the soils at the study site were high hydraulic conductivities with associated risk of leakage to Red House and Anvil Creeks, low nutrient retention and locally high antecedent soil moisture content with associated reduction in permeability (HWC, 1991). From 1991 to 1995 the HWC implemented woodlot assessments and monitoring schedules for groundwater, effluent and soil properties, the results of which are reported in HWC Interim Progress Report (1991), Dennis and Sinclair (1994) and HWC Progress Report 3 (1995).
The monitoring reports present results from a preliminary report (HWC, 1987) and annual soil sampling, monthly groundwater chemistry, tree growth and performance comparisons and hydraulic loading/volumetric soil moisture data (HWC, 1991; Dennis and Sinclair, 1994; HWC, 1995). Very little monitoring occurred after 1995 although it recommenced with the start of this project in December 2001.

Relevant results from these reports have been summarised in Section 3.7. In addition, the Branxton Golf Course and an adjacent grazing property have irrigation priority to the woodlot studied, thus during drier times, the woodlot received inadequate irrigation. Sampling sites from previous monitoring, discussed in Section 3.5 and 3.7, are also shown in Figure 3.2.

![Figure 3.1: Location of study site](image)
Figure 3.2: (a) General layout of the Branxton Waste Water Treatment Works (WWTW) and irrigated woodlots and (b) Area 3 showing past monitoring sites (groundwater bores and soil sampling sites (4, 6 and 7)) and treated areas (lime addition)
3.3 Tree species planted

Area 1 (1.12 ha) and Area 3 (1.32 ha) are shown in Figure 3.2, although Area 3 was the only site monitored for this study. Area 2 was a proposed woodlot site deemed unsuitable due to shallow topsoils (HWC, 1991) and is not shown. In 1992, two trial plots (Area 1 and Area 3) were planted with *Eucalyptus* species (predominantly); with Area 1 irrigated by a drip system and Area 3 irrigated with a sprinkler system (Figure 3.2).

Area 1 has several species of trees including *Eucalyptus grandis*, *Acacia melanoxylon*, *Casurina cunninghamiana* and *Eucalyptus saligna*. Since potential evapotranspiration was a significant water balance parameter to be monitored and different trees have different water uptake characteristics, it was considered appropriate to study an area that consisted of a single species. As a result, Area 3 was selected as the site for this study and is dominated by *E. grandis*. Every second row in Area 3 was coppiced in late 1996 to remove lesser quality trees and to thin the woodlot canopy. By 2003, the height of non-coppiced trees ranged from approximately six to 11 metres and the height of coppiced trees ranged from approximately, one to five metres. Approximately 200 trees exist in the studied woodlot.

3.4 Climate

Annual temperature for Branxton ranges from 4 – 44°C. The annual average rainfall for Branxton has varied between 650 – 850 mm for the last 50 years and falls mainly in autumn and winter (April – June) (McManus et al., 2000). Rainfall is usually generated by high-pressure systems that intensify inland and produce rain on meeting cooler maritime air (McManus et al., 2000). The intensity of precipitation during the study period has been recorded as high as 175 mm in 24 hours (personal observation, February 2002). The frequency and intensity of these storm events varies from year to year, although a relationship between the Southern Oscillation Index (drier/wetter periods) and annual rainfall has been recognised in the Branxton area (McManus et al., 2000), which may be of use for future modelling and rainfall prediction.
3.5 Groundwater

The locations of groundwater bores used in previous monitoring are shown in Figure 3.2b. From 1991 to 1995, HWC initiated groundwater sampling and analyses which revealed that EC ranged from approximately 300 µS/cm to 7000 µS/cm, whilst the pH for all boreholes was relatively stable and held a range of 5.5 – 7.0 over the same period (Dennis and Sinclair, 1994). Groundwater depth from 1991 to 1995 was measured every two months and ranged from 0.2 m to 2.5 m for all boreholes and was of low turbidity.

As used in this study, the long-term measurement of groundwater depth was an important indicator as to whether the aquifer was recharging, stable, or in decline. For example, a rising groundwater level indicated that the aquifer was being recharged and susceptible to over-irrigation. By observing changes in groundwater depth over time and determining groundwater flow direction and velocity in relation to hydraulic loading, it was possible to identify the most responsive boreholes to hydraulic loading, whose levels could act as indicators for potential problems.

For example, if the topographically lowest borehole was less than 0.4 m below ground level, it was highly probable that saline groundwater will emerge at the topographically lowest point between the woodlot and Anvil Creek. This point exists on an adjacent property and there is evidence that saline scalding has occurred in the past (Dalby and Geary, 2000). Site groundwater is predominantly saline and can exist close (< 0.5 m) to the surface at the lower western corner of Area 3 (Dennis and Sinclair, 1994). In addition, several dead or dying Yellowbox trees (E. meliodora) are present in the adjacent property, which is predominantly used for grazing.

3.6 Geology and soil description

The soils at the Branxton site are predominantly duplex (Figure 3.3), with loamy sands/sandy loams overlying clay at a depth of approximately 35 cm. A previous geotechnical report (HWC, 1987) found that sandstone bedrock exists at a depth of approximately 90 cm near the WWTW entrance (elevation = 40 m AHD), which dipped
The sub-surface clay was found to extend beyond the excavation depth of 1 m.

![Diagram of soil profile]

The duplex soil in Area 3 has a clear and abrupt textural B horizon boundary where the upper 0.2 m of the B2 horizon is sodic and weakly sub-plastic. Using Isbell’s classification (1996), the Area 3 woodlot soil would be classified as a sodosol. With the dominant colour being brown, a mottled B horizon with an ESP > 25 and a Ca/Mg ratio less than 1, this soil can be further classified into sub classes, resulting in a Brown (AB), Mottled – Hypermatric (FP), Magnesic (DB) Sodosol.

Using the Northcote Factual Key (Northcote, 1971), the soil is classed as Db4.41; a duplex soil with a brown (predominantly, with some yellow) clay B horizon, A horizons that are non-hardsetting, a conspicuously bleached A2 horizon where the surface soil pH is less than 7 and the sub-surface soil pH is less than 6.5. A combination
of both soil classifications provides a detailed description of the soils in Area 3 at the Branxton WWTW effluent irrigated woodlot.

3.7 Soil ESP and hydraulic loading from past monitoring.

Figure 3.4 shows soil ESP between 1991 and 1995 for each site previously monitored by HWC (HWC, 1991; Dennis and Sinclair, 1994; HWC, 1995). Only soil sample sites that existed within Area 3 are shown. The suffix “A” or “B” represents the 0 – 10 cm and 30 – 40 cm depths respectively. Sites 4 to 7 (note: no sample was taken at site 5) are situated on alluvial sand deposits, although site 7B was never sampled. Sites 4, 6 and 7 were shown in Figure 3.2.

Soil ESP appears to have increased over time according to Figure 3.5. Site 4A experienced increased ESP from approximately 3 in 1991 to 14 in 1995, while the subsurface 4B increased from approximately 2.5 in 1991 to 10 in 1995. Site 4B did have a higher ESP in 1994 than in 1995 (1994, ESP was approximately 13). Site 6A experienced the largest increase, with ESP values ranging from approximately 5 in 1991 to 30 in 1995. Soil ESP values for 6B ranged from 5 in 1991 to 27.5 in 1995. Site 7A had increased from approximately 2 in 1991 to approximately 7 in 1994 and 1995.

![Figure 3.4: Temporal soil ESP at selected sites (4A, 4B, 6A, 6B and 7A) within Area 3 (from past reports)](image-url)
Sites 4 and 7 are located in areas within the woodlot comprising loamy sand textures to a depth beyond 1 m. Due to the textural contrasts, both would be expected to have a relatively higher permeability than Site 6. Site 6 has sandy clay at a depth of approximately 40 cm, and due to increased clay content would be expected to have a relatively lower permeability. Figure 3.5 shows the annual depth of rainfall and effluent applied to the 1.32 ha woodlot (Area 3) between 1992 and 2003 and the percentage of the total effluent generated by the WWTW that was applied to Area 3 (1.32 ha).

Irrigation of secondary treated effluent increased between 1992 and 1994, meaning the site received increased sodium loading as well. For several years (1994 – 1999), effluent application remained relatively high (1800 – 2550 mm) then decreased from 1999 onwards. Rainfall during 1995 and 1996 was at approximately double the average of the previous three years and represent the wettest years in the dataset.

The total effluent generated for each year was based on 459 ML/yr inflow to the WWTW. However, in reality some will be lost to evaporation and pond leakage, although these losses were not incorporated into Figure 3.5. The trend in the percentage of the total effluent generated by the WWTW decreases in the early years and slowly increased from 1997 onwards, with the woodlot receiving lower irrigation volumes in 2002/03.
Chapter 4: METHODS

4.1. Introduction

Site-specific water balance components, volumetric soil moisture, secondary treated effluent chemistry and soil chemistry were all monitored from January 2002 – October 2003 and used to determine the temporal sodium flux in a woodlot soil receiving secondary treated effluent. This chapter describes the methods used to acquire data to accomplish the aims of this thesis. The approach taken and methods used are summarised in Figure 4.1.

Aims 1 and 2 of this research involve monitoring the site water balance, soil chemistry and volumetric soil moisture to examine the temporal sodium accumulation/leaching (flux) in a woodlot soil receiving secondary treated effluent. Aim 3 was to investigate the extent to which variations in soil ESP and effluent SAR over time impact upon soil structure, in order to better manage STE irrigation scheduling and soil structure. Methods for achieving Aims 1 and 2 are discussed in Sections 4.2 and 4.3. Methods for achieving Aim 3 are discussed in Section 4.4.

4.2 Field methods

4.2.1 Measurement of water balance components

To avoid surface runoff and/or prolonged surface ponding, irrigation must not exceed the daily volume evaporated and transpired by the woodlot. In this study, potential evapotranspiration is equivalent to evaporation and transpiration by the woodlot. Therefore the irrigation scheduling implemented in this study was calculated based on daily potential evapotranspiration, surface runoff, deep drainage, interception loss and applied effluent/rainfall:

\[ \Delta I = [PET + R + D - ((Qp + Qe) - IL)] \]

(Equation 11 from Chapter 2)

where \( \Delta I \) = irrigation deficit/surplus, PET = Potential evapotranspiration (determined from evapotranspiration (ET) calculations), R = water lost as surface runoff and D = water lost as percolation and/or deep drainage, IL = interception loss, Qp = precipitation and Qe = applied effluent. All parameter units are in millimetres.
FIELD METHODS

Water Balance Component
- Effluent + rainfall loading
- Evaporation + transpiration (as PET);
- Interception loss, runoff and deep drainage;
- Daily irrigation deficit/surplus
- Cumulative daily irrigation deficit/surplus;
- Volume-weighted average SAR
- Groundwater (flow rate and direction)
- Volumetric soil moisture ($\theta$)

Soil Component
- Soil sampling
- Logging of sample sites
- Bulk density
- Sodium loading from irrigation
- Site observations with respect to surface ponding after rainfall and/or irrigation
- X-ray diffraction (XRD) of site soils to identify dominant clay types

LABORATORY METHODS
Effluent, rainwater and groundwater analyses: Electrical conductivity, pH, calcium, magnesium, sodium and potassium and Sodium Adsorption Ratio (SAR).

Soil analyses: Electrical conductivity, pH, Cation Exchange Capacity, Exchangeable Sodium Percentage (ESP).

Soil pressure plate extraction methods: to determine field capacity (FC) and wilting point (WP) from soil moisture curves.

Column Leaching Experiments (CLE): to monitor change in permeability of the studied soils, including C/C, solute breakthrough curves inferring advection/hydrodynamic dispersion processes.

SOIL WATER (SOLUTION) Interactions SOIL (EXCHANGE COMPLEX)

FIGURE 4.1: Methods summary flowchart.
In this study, cumulative irrigation surplus/deficit was a limiting component for
daily irrigation scheduling and was used to identify probable periods of sodium
accumulation and/or leaching over time and depth, based on the measured volumetric
soil moisture ($\theta$) at the time of irrigation (discussed in Section 4.2.9). The daily
irrigation deficit/surplus was designed to be zero and was arbitrarily chosen at a time
when effluent irrigation was not possible for several weeks due to observed site
saturation, for example, in February 2002. The soil was deemed “saturated” at this time
in terms of further effluent application.

4.2.2 Applied effluent (Qe) and Precipitation (Qp)

Applied effluent volumes were recorded daily from pump readings. The
automatic sprinkler system was usually set on a weekly irrigation schedule that could be
altered daily if required. For example, if a large rainfall event occurred the previous
day/night, the system could be halted or reduced so as to not exceed PET and/or
recharge the soil water storage for the following day. Effluent samples were collected
monthly from a licensed EPA discharge outlet and placed in cool storage (< 6 °C) until
arrival back at the laboratory.

Precipitation depth was recorded daily using a rain gauge and plotted to show the
frequency of rainfall over time at the study site. Precipitation (Qp) was sampled using a
clean 15-litre bucket placed on the ground during an intense rain event. Both the STE
and rainwater samples were analysed as shown in Table 4.1 in Section 4.3.1.

4.2.3 Evapotranspiration (as PET)

Potential evapotranspiration (PET) was determined from data acquired from a
SensorMonitor™ Class “A” pan multiplied by a pan co-efficient ($K_p$) (Myers et al.,
1992). The pan was installed to the west of the Pasveer oxidation ditch at the Branxton
WWTW (see Figure 3.2). This data logging auto refill instrument was mounted on a
concrete slab with hoses and cables installed underground, as the unit uses both power
and water. A protective mesh covered the pan to prevent wildlife drinking the water.
Daily data were downloaded on a weekly basis and used to estimate the ET of *E. grandis* in the woodlot studied. To determine ET, E was multiplied by K_p, which represented an estimate of woodlot transpiration (T). No direct measurement of transpiration was made in this study. The pan coefficient (K_p) for tree water use was assumed to be greater than 1, as trees have higher water uptake potential (larger root zone and leaf surface area) than pasture, where pan coefficients have been noted as being less than 1 (Brown *et al*., 2001).

The pan co-efficient has been shown to vary throughout the seasons (Honeysett *et al*., 1992; Myers *et al*., 1999) and this variation has been incorporated into the water balance used. Since PET equals ET in this study and as no direct measurement of transpiration (T) was made, PET was defined as:

\[(\text{PET}) = (E) \times K_p\]  
(Equation 12)

Monthly K_p values ranged from 1.09 to 1.43 and were taken from data by Myers *et al*., (1999). Incorporated in the water balance calculation in Equation 11, daily PET values were used over the study period to establish the loss of water attributed to woodlot evapotranspiration. Variations in monthly K_p were based on data from Myers *et al*., (1999).

### 4.2.4 Interception loss (IL)

Dependent on canopy cover and density of understorey, interception loss (IL) was estimated. Interception loss also depends on the duration and intensity of the rainfall event and varies accordingly. For example, if rainfall is less than 5 mm/day, IL will be close to 100% because light rains are absorbed or captured, which prevents water reaching the ground (Myers *et al*., 1999). In contrast, during intense rainfall, IL can be as little as 5-10%. For a flooded gum (*E. grandis*) plantation at Wagga Wagga, IL was found to be 12% in winter and 18% in summer (Myers *et al*., 1999).

In this study, IL was estimated using the above values from Myers *et al*., (1999) as a guide only. The canopy coverage was estimated at 50% of the canopy coverage of the Wagga Wagga site. No measurement of interception loss was made and since the woodlot contains 50% mature trees and 50% coppiced trees, the IL values used in this
study were half the values used at Wagga Wagga. The IL values used were 6% for winter/spring months (October - March) and 9% for summer/autumn months (April - September).

4.2.5 Runoff (R)

This component becomes increasingly important as ΔI increases although R is usually designed and/or estimated to be zero with respect to applied effluent. Due to the low slope of the study site, visible surface runoff was not observed during the wettest periods of this study. However, due to intense rainfall, prolonged surface ponding did occur on occasions and resulted in the halting of effluent irrigation to the site. The component, R, was assumed to be zero with respect to water calculations in Equation 11, although any positive daily irrigation surplus resulting in prolonged surface ponding can be attributed to this component. However, surface runoff was assumed to be zero throughout the study period, in order to highlight surplus events that satisfied PET and available soil water storage. The coarse textured soil type of the woodlot, the pervious distances from adjacent properties and lack of observed runoff meant that runoff is most likely to move subsurface and most likely directed through the drainage line.

4.2.6 Deep drainage (D)

Deep drainage (D) depends on soil permeability, timing and volume of rainfall/irrigation and the volumetric soil moisture condition (θ) during rainfall and/or effluent application (Myers et al., 1999). In the design of an effluent irrigated woodlot, it is common to include a Leaching Requirement (LR) to assist in removing salts from the woodlot root zone (Snow et al., 1999) which in effect is replaced by D in water balance calculations (Equation 11).

The component, D, was initially assumed to be zero with respect to water calculations in Equation 11, although any negative daily ΔI resulting in zero surface ponding can be attributed to this component. Since D was not actually measured, the comparison of hydraulic loads and subsequent increases in groundwater levels were also used to determine if deep drainage was occurring, with the assumption that if
Groundwater levels are rising then applied waters and rainfall are being transmitted throughout the depth of the soil profile, thus deep drainage was occurring.

4.2.7 Groundwater

The variation in groundwater depth over the study area was determined by the use of groundwater bores. Five groundwater bores were installed and surveyed in March 2002 around Area 3, although shortly after installation one of them (GW4) became blocked and remained so for the duration of the study period. Because of funding restrictions it was never repaired.

All bores were installed according to EPA (1995) guidelines, with each consisting of a gravel-lined, slotted (screened) 50 mm PVC pipe. Groundwater depths were measured weekly using a “fox-whistle” attached to an incremented chain. The “fox-whistle” is a hollow steel tube designed to whistle when it reaches groundwater surface. The depths recorded have an error of ± 0.05 m, which was one-half of the smallest chain increment. Groundwater samples were collected monthly using a water displacement tube, which enabled sampling of the water column.

Since trends in sodium accumulation/leaching were to be determined, groundwater hydraulic conductivity \((K)\) under the effluent irrigated site was also obtained. Groundwater \(K\) was determined using a “slug” injection technique, based on Hvorslev (1951). A “slug” is an addition of water to the borehole aimed at increasing the hydraulic head. This method states that flow from a well, “\(q\)”, at any time “\(t\)” is proportional to the \(K\) of the aquifer and to the decline of the head level in the well. By neglecting the storage effect of the aquifer, the \(K\) of the aquifer around the slug test portion was estimated by:

\[
K = 2.303 \frac{A}{F} \left[ \frac{\log(H_1/H_2)}{t_2-t_1} \right]
\] (Equation 13)

where \(A\) is the area of the borehole (mm\(^2\)), and \(H_1\) and \(H_2\) (mm) are the heads recorded at times \(t_1\) and \(t_2\), respectively (Mas-Pla et al., 1997). \(F\) is a shape factor, which depends on the geometry of the well intake and its location within the aquifer (Mas-Pla et al., 1997).
The shape factor chosen for this study corresponded to the case of a point piezometer, as illustrated in Figure 4.2, while the set-up of the borehole is shown in Figure 4.3.

![Figure 4.2: A case of a point-piezometer (after Hudak, 1999)](image)

The shape factor, $F$, was expressed as:

$$ F = \frac{2\pi L}{[\ln(L/2r_w) + [1 + (L/2r_w)^2]^{1/2}]} $$

(Equation 14)

where $L$ is the length of screen and $r_w$ its radius. This formula assumes that the length of the screened interval of the slug test is one-tenth of the saturated thickness of the aquifer, and the flow regime resulting from the slug test is symmetrical about a horizontal plane through the centre of the well.

The pressure head ($h_p$) at the bottom of the pipe is equal to the length of the water column in the pipe. The elevation of the bottom of the pipe is the elevation head ($h_e$). The sum of both gives the hydraulic head ($H$), whilst the screened interval is the slotted length of the piezometer that allows groundwater to flow freely through the pipe.
A slug was added to one of the boreholes (6/6/03) and the head level measured every third day for 14 days. At this time, groundwater levels were stable for several readings and the test ended.

Groundwater direction and magnitude were determined using the 3-point method as described by Hudak (1999). The method involves selecting at least three borehole sites from a scaled map and drawing a triangle between the points (refer to example in Figure 4.4). The side connecting the highest and lowest water levels were then selected (A to B). A point was found on this line that had a water level elevation equal to the third borehole (point X, equipotential to C).

A line is then drawn between the third borehole and the showed point on the line that joins the highest and lowest water levels (C to X). The direction of groundwater flow is perpendicular to that line, in the direction of the decreasing head (from A to line BC).
By drawing the perpendicular to the point with the lowest water level elevation, the magnitude of the hydraulic gradient can be estimated. Measured in the direction of groundwater flow, the hydraulic gradient is the difference of the hydraulic head divided by the distance between two points. The hydraulic gradient is obtained, as a dimensionless unit, by subtracting the lowest water level from the intermediate water level and dividing that difference by the length of the perpendicular (refer Figure 4.4).

**4.2.8 Soil component - site selection and soil sampling**

In this thesis, the soil sampling strategy required to determine the sodium flux consisted of:

- The establishment of a pattern of sampling points based on variations in soil type and textural boundaries;
- Sampling soils at 0.1 – 0.2 m depth increments to better define vertical trends over time;
- Increasing the sampling frequency to provide greater insight into temporal sodium flux.
The aim of soil sampling and volumetric soil moisture access tube site selection was to obtain a representative sample of the Area 3 woodlot (1.32 ha). A 50 mm manual soil corer was used to sample surface and sub-surface horizons. During installation of access tubes for volumetric soil moisture measurement, soil profiles were logged and increments of 20 cm were sampled. In total, 14 samples were taken (12 inside the woodlot and two outside (non-irrigated sites)) on the 18/12/01 and 20/2/02 (refer Figure 4.5). The 0-10, 10-20, 20-40, 40-60, 60-80 and 80-100 cm increments from inside the woodlot were each combined, with the aim of providing a representative bulk sample (by depth increment) for use during column-leaching experiments and to measure soil moisture curves.

Four sites within the woodlot (S1, S5, S8 and S10) and two sites outside the woodlot (S9 and S14) were selected for ongoing monitoring and sampling. On each sampling occasion, the soil sample was taken within a two-metre radius of the initial site position. After sampling, soils were sealed in plastic bags to minimise moisture loss. Soils were sampled every two months from February 2002 to October 2003 with Figure 4.5 showing all selected monitoring sites in Area 3. Sample holes were back-filled with dry soil. The dry back-fill soil (non-irrigated) was previously sampled from an area adjacent to the woodlot for this purpose.

Each soil site selected for long-term monitoring, namely S1, S5, S8, S9, S10 and S14 was assumed to adequately portray the variability of textural classes at specific sites within Area 3. The reason for two non-irrigated sites was that S9 has clay at + 40 cm, while S14 has no clay at all to the sampled depth of 1 m. The woodlot consisted of approximately 50% soil similar to S9 and 50% soil similar to S14.
Figure 4.5: Area 3 woodlot - soil sampling sites, soil moisture sampling sites and groundwater bore locations
4.2.9 Volumetric soil moisture ($\theta$)

Volumetric soil moisture content ($\theta$) was monitored weekly using a Gopher Soil Moisture Profiling System. This system was selected because of its capacity to generate similar results to neutron probe methods, although at a reduced cost (Grabham, 1999 (unpublished report)). The purpose of monitoring volumetric soil moisture was to assess weekly net changes in soil moisture, which were used in water balance calculations to obtain a cumulative irrigation surplus/deficit trend.

The Gopher system uses a capacitance probe to measure volumetric soil moisture at various depths within a soil profile (Figure 4.6). The 50 mm access tubes are constructed of PVC and sealed at one end before installation (Dataflow, 1999). Installation of the access tubes was achieved using a 46 mm soil corer to excavate the holes.

![Figure 4.6: Gopher soil moisture profiling system](image)

The corer provided with the Gopher system (tube cutter type) was used to smooth the sides of each hole. The access tubes were then pushed into the holes to ream the sides and provide a tight fit. The staff, with capacitance probe attached, was then placed into each access tube to ensure that the tubes were not warped during installation.
Fourteen 50 mm PVC access tubes were bored to a depth of 0.8 m in Area 3 at the study site (12 in the woodlot and two outside the woodlot). The capacitance probe, mounted on an incremented staff, was inserted into these access tubes. The top of the staff was connected to a hand-held data logger, enabling soil moisture values to be obtained over a number of depths each site visit.

Calibration of the Gopher probe was achieved in the field. Saturated soils in February 2002 allowed the Gopher to be calibrated in-situ, however the handbook “soil type” used was the “loamy sand” (for A horizons) and “clay” (for B horizons) which was selected from the given textures in the Gopher software (which coincided with the description of site soils). Saturated soil moisture values ranged from 36 mm to 40 mm in the A horizon based on 15 readings at S8. Saturated soil moisture values ranged from 55 mm to 58 mm in the B horizon based on 15 readings at S10. All A horizons were assumed to have similar hydraulic properties throughout the 1 ha woodlot, as were all B horizons. Due to textural contrast, A horizons (loamy sand) were assumed to have greater hydraulic conductivities than B horizons (sandy clay).

Gravimetric soil moisture \( \theta_g \) was also determined from the soils sampled every two months to validate Gopher volumetric soil moisture data over the long-term. Soil sampling was performed as discussed in Section 4.2.8 with the volumetric soil moisture \( \theta \) being determined by not only direct measurement using the Gopher Soil Moisture Profiling System, but also the conversion of \( \theta_g \) to \( \theta \) by the equation:

\[
\theta = \theta_g \times BD
\]

(Equation 15)

where BD = bulk density (g/cm\(^3\)).

4.3 Laboratory methods

4.3.1 Effluent, rainwater and groundwater analyses

All water samples taken during the study period were analysed using methods from Standard Methods for the Examination of Water and Wastewater (APHA, 1995). Table 4.1 summarises the standard methods used for the analysis of waters collected during this research.
Table 4.1: Methods for Water Analyses (effluent, rainfall and groundwater)

<table>
<thead>
<tr>
<th>PARAMETER</th>
<th>METHOD</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH</td>
<td>APHA (1995) 4500–H⁺ B</td>
</tr>
<tr>
<td>Electrical Conductivity (EC)</td>
<td>APHA (1995) 2510 B</td>
</tr>
<tr>
<td>Calcium (Ca²⁺)</td>
<td>APHA (1995) 3–56</td>
</tr>
<tr>
<td>Magnesium (Mg²⁺)</td>
<td>APHA (1995) 3–75</td>
</tr>
<tr>
<td>Sodium (Na⁺)</td>
<td>APHA (1995) 3-96</td>
</tr>
<tr>
<td>Potassium (K⁺)</td>
<td>APHA (1995) 3-82</td>
</tr>
<tr>
<td>Turbidity (NTU)</td>
<td>APHA (1995) 2130 B</td>
</tr>
<tr>
<td>Sodium Adsorption Ratio (SAR)</td>
<td>Calculation using Equation 2</td>
</tr>
</tbody>
</table>

pH was measured using a Eutech Ecoscan pH5 instrument that was calibrated prior to use with pH buffers of 4.00 and 7.00. The EC was measured using a Eutech Ecoscan Con5 instrument that was calibrated prior to use with a 1413 μS/m (0.01M) potassium chloride solution. Turbidity was determined using a Lovibond Turbidity Meter that was calibrated using 10, 100 and 1000 NTU formazin solutions.

After filtering and acidification, calcium (Ca²⁺), magnesium (Mg²⁺), sodium (Na⁺) and potassium (K⁺) were determined using an Australian Research Laboratory 3520 inductively coupled plasma – atomic emission spectrometer (ICP-AES). The calibration series consisted of standards containing 0, 25 and 50 mg/L of each cation. Each sample “run” on the ICP-AES comprised of three readings, which were presented as an average. Results that exhibited a relative standard deviation (RSD) of > 2 % for the three readings were re-analysed until the RSD was less than 2 %. Sample results exceeding the maximum calibration standard were diluted accordingly and re-analysed.

4.3.2 Soil analyses

Analyses began immediately upon return to the laboratory, where samples were weighed, dried at 40°C for 48 hours and then re-weighed to obtain the gravimetric soil moisture content (θₑ). Each sample was then sieved at 2 mm to yield ≤ 2 mm and > 2 mm fractions, a common size for soil extraction methods (Rayment and Higginson, 1992). The > 2 mm was crushed to approximately 2 mm with a Jacques jaw-crusher and
re-sieved at 2 mm. The ≤ 2 mm then became the prepared sample for analysis. The soil analysis methods are summarized in Table 4.2.

**Table 4.2: Soil analysis methods (after Rayment and Higginson, 1992)**

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Method</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>pH in soil</td>
<td>4A1</td>
<td>1:5 in distilled water, tumbled for 1 h, left to rest for 20 minutes; then the supernatant analysed using a pH meter.</td>
</tr>
<tr>
<td>EC in soil</td>
<td>3A1</td>
<td>1:5 in distilled water, tumbled for 1 h, left to rest for 20 minutes; then the supernatant analysed using an EC meter.</td>
</tr>
<tr>
<td>Exchangeable Cations (Ca, Mg, Na, K)</td>
<td>15A1</td>
<td>5g in 100 mL. Tumbled for 1 h in 1M NH₄Cl adjusted to pH7, filtered through Whatman No 54 filters then analysed using ICP-AES.</td>
</tr>
<tr>
<td>Exchange Acidity (Al³⁺ and H⁺)</td>
<td>15G1</td>
<td>8g in 80 mL. Tumbled for 1 h in 1M KCl, filtered through Whatman No 2 and titrate with 0.02M NaOH.</td>
</tr>
<tr>
<td>Cation Exchange Capacity (CEC)</td>
<td>15N1</td>
<td>Calculated as Exchangeable Cations + Exchange Acidity</td>
</tr>
<tr>
<td>ESP</td>
<td>15N1</td>
<td>Calculated from Equation 3 (Chapter 2)</td>
</tr>
</tbody>
</table>

Soil samples were analysed using methods from Rayment and Higginson (1992) and included pHₕₑ (Method 4A1), electrical conductivity (Method 3A1 – in dS/m), exchangeable cations (Method 15A1 - Ca²⁺, Mg²⁺, Na⁺, and K⁺ in meq/kg), exchange acidity (Al³⁺ + H⁺ in meq/kg) and exchangeable sodium percentage (Method 15N1 - calculated from Cation Exchange Capacity).

For determination of exchangeable cations, Rayment and Higginson (1992) infer that pre-treatment of soils with soil EC’s less than 0.3 dS/m is not required. The majority of soils sampled over the study period had a soil EC less than 0.3 dS/m (87% of all samples). Furthermore, since no pre-treatment was required it was assumed that the sum of Ca²⁺, Mg²⁺, Na⁺, and K⁺ equalled exchangeable cations and not extractable cations. The methodology used for determining exchangeable cations for soils with EC values < 0.3 dS/m does not allow for differentiation between exchangeable/extractable cations.
Based on this the exchangeable cations which equals the sum of Ca$^{2+}$, Mg$^{2+}$, Na$^+$, and K$^+$ was used in soil ESP calculations.

To determine the significance of not pre-treating soil samples with soil EC’s > 0.3 dS/m, exchange acidity (Al$^{3+}$ and H$^+$) was determined from selected samples. Exchange acidity was determined for 20 selected samples based on ranked pH. All soil samples were ranked from lowest to highest pH and 20 of the top 50 (covering a range of approximately 0.9 pH units) were representatively selected based on their depth, texture and location in the woodlot. The selected sites were representative of the soil profiles studied and included approximately equal proportions of A and B horizons with both loamy sand and sandy clay textures.

Exchange acidity significantly decreases as pH increases (White, 1997) and therefore the significance of exchange acidity in CEC calculations also decreases as pH increases. The decreasing Exchange Acidity versus increasing pH relationship, and also using the comparison of CEC values (with and without inclusion of exchange acidity), justifies the use of exchangeable cations as a surrogate for CEC. This is also validated by the relatively high pH range and relatively low CEC values of the sampled soils.

### 4.3.3 Calculation of the sodium loading

Salts are added to woodlot soils through effluent irrigation, which is usually reflected in the electrical conductivity of the soil. Assuming negligible sub-surface lateral movement, the quantity of salt that accumulates in the soil will depend on the sodium applied from effluent and rainfall, exchangeable bound sodium in the soil (expressed as an ESP value), the amount lost to lateral movement and the quantity leaving the root zone by leaching. The quantity taken up by trees is negligible (Taylor, 1996). Sodium pathways are represented in Figure 4.7.
Previous research has used the EC of the soil to determine “salt” loading (Myers et al., 1999), whereas this research used actual cation concentrations and correlated these to predicted micro-aggregate/soil pore stability of the site-specific soil. Once the volumes of applied water (irrigation/rainfall) were determined, it became a matter of mass balance to determine sodium flux.

To calculate the sodium loading for each monitoring period, the following equation was used:

\[
\text{Applied Mass of Na}^+ = \text{volume of effluent applied} \times \text{effluent Na}^+ \text{ concentration} \\
\text{(mg)} \quad \text{(L)} \quad \text{(mg/L)} \\
\text{(Equation 16)}
\]

Conversion of milligrams to grams to kilograms was also performed where appropriate. Substituting rainfall for effluent into the above equation defined the mass of sodium contained in any given rain event. The change in soil ESP between a given period shows the net loss/gain in sodium with respect to the water balance and applied
effluent. For example, although represented as a percentage, the soil ESP actually describes the relative amount of sodium in cmol(+)/kg (molar concentration). Therefore, any increase or decrease in soil ESP between sampling occasions represents a significant change in the presence of sodium, relative to calcium and magnesium.

To calculate the volume-weighted SAR of the total applied water (effluent + rainfall) for any given day, a simple mixing-model was used which is represented by the following equation:

\[
netSAR = \frac{C_eV_e + C_pV_p}{V_e + V_p}
\]  

(Equation 17)

where \(C_e\) is the SAR of applied effluent, \(V_e\) is the volume of effluent applied, \(C_p\) is the SAR of rainfall and \(V_p\) is the volume of rainfall. For example, if 20 mm of effluent (SAR = 10) was applied and 10 mm of rain also fell (SAR = 0.6) on a given day, the resulting SAR of the total applied water would be 6.9 (1 decimal place) for that day.

Since soil permeability is a function of both soil ESP and effluent SAR, this study looked at sodium loading in two ways. Firstly, the amount of sodium added through effluent application (in kg), and secondly, the relationship between sodium and other cations in solution (as an SAR value). In addition, when the trees’ water needs exceed the applied water, it was assumed that sodium was accumulating, whilst it was assumed that sodium was downwardly mobile in a saturated soil profile.

4.3.4 Soil bulk density

Bulk density was obtained by using a metal ring of known volume, which was inserted into the ground by sledgehammer, then removed to acquire an undisturbed soil core. The soil contained in the ring was removed by hand and then double-bagged and sealed to prevent moisture loss. In the laboratory, the moist soil weight was recorded (in g). The soil was then placed in a 40°C oven for 48 hours then re-weighed to attain a dry mass (in g).

Four samples using a sampling grid were taken on two occasions (6/7/2002 and 25/4/2003) throughout Area 3 and the calculated bulk density values averaged to gain a representative bulk density. Also, the bulk density variance (95% confidence interval) of
the woodlot was calculated using the Student’s $t$-distribution and the equation (Parl, 1967):

$$
\mu = x \pm 2.14 * \frac{s}{\sqrt{n-1}}
$$

(Equation 18)

where $\mu$ is the 95% confidence interval, $x$ is the sample mean, $s$ is the relative standard deviation and $n$ is the number of samples.

The bulk density was then calculated from the equation:

$$
BD = \frac{mass(kg)}{volume(m^3)}
$$

(Equation 19)

where mass (g) is the air-dry mass at 40°C.

4.3.5 Soil moisture curves

Soil moisture curves were developed to determine wilting point and field capacity of the studied soils. Soil potential graphs were generated for Branxton woodlot soils between 0.1 and 10 kPa using the soil moisture extraction plate method, similar to that used by Barlow and Nash (2002). Ten kilopascals was used as wilting point in this thesis when in fact it should have been 15 kPa. Each sample analysis was conducted in triplicate at each designated pressure, using the 0 – 20 (a combination of 0-10 and 10-20 cm samples), 40 - 60 and 80 - 100 cm soil depth fractions. Pressure levels used were 0.1, 0.3, 0.5, 0.9, 1.2, 3, 5 and 10 kPa. The sample used for this test was the bulk representative soil, which was previously sampled and prepared as discussed in Section 4.2.8. Limitations of this approach are discussed in Chapter 6.

To construct soil moisture curves, soil moisture extraction tests were performed which involved placing saturated soil slurries into PVC rings (radius = 25 mm, depth = 6 mm) into a pressure plate extractor (Cat. No. 1500, Soilmoisture Equipment Corporation, USA). Soil characteristic curves at pressures of 0.1 – 10 kPa were determined using a 100 kPa ceramic pressure plate (Cat. No. 1290, Soilmoisture Equipment Corporation, USA) connected to a suction system. This was repeated for successive pressures.
After 48 hours in the pressure plate extractor, samples were removed, weighed and placed in a drying oven (40°C). Samples were weighed to constant-mass (< 0.4 g / 24 hours) and the volumetric soil moisture determined by using Equation 14. From the graphs created, it was possible to define both wilting point (WP) and field capacity (FC). FC was recorded from the y-axis at a pressure of 0.3 kPa and WP taken at 10 kPa.

The Gopher measurement (volumetric soil moisture, $\theta$) taken on a weekly basis in situ gave trends in soil moisture (increases/decreases) over time. Soil retention curve values were multiplied by bulk density in order to compare Gopher Soil Moisture results and to recognise optimum soil moisture (wilting point and field capacity) for the planted species (*Eucalyptus grandis*) at the Branxton site.

4.3.6 Column-leaching experiments

**Column-leaching in relation to the $C_{TU}$ and $C_{TH}$**

Soil in irrigated woodlots undergoes changes in soil structure when the SAR of the application waters varies between the $C_{TU}$ and $C_{TH}$. An experiment was undertaken to illustrate the impact of varying SAR on soil structure by observing the presence of dispersed clay particles in the output solution, after equal volumes of permeants had been applied. The concept was to subject the same soil to the extremes in SAR values brought about by effluent irrigation and rainfall. To establish some degree of consistency between laboratory experiments and field conditions, the packing of the columns was representative of in situ soils. Therefore, bulk density values determined from field site sampling were used as a surrogate to establish column-packing specifications.

In the first set of experiments, two columns (1 and 2) were packed with the < 2 mm size fraction of the 0 – 10 cm depth increment of soils from the site, to the average bulk density value calculated previously. The column was then tapped lightly until the depth of soil was 10 cm, thus representing a disturbed soil column for the 0 – 10 cm depth increment. Each column was packed lightly with a small piece of glass wool to prevent surface soil displacement during permeant application. All soil samples taken during site monitoring were prepared according to Australian Standard AS1479.1-3 and analysed at less than 2 mm using methods from Rayment and Higginson (1992). Soil
samples were analysed for exchangeable cations (Ca$^{2+}$, Mg$^{2+}$, Na$^+$, K$^+$) in each horizon (Table 4.2), prior to placement in columns.

Column 1 and Column 2 both had soil of ESP = 21.6, which was determined from the bulk soil was analysed prior to use. Column 1 was applied with 500 mL of secondary treated effluent (SAR = 5.2) and Column 2 was applied with 500 mL of rainwater of SAR = 0.5. The volume of solution applied and the time taken to permeate through the column was recorded. Visual observations were also made and noted regarding turbidity and the presence of clay particles in the water eluted from the columns.

**Column-leaching by depth**

Column experiments were performed in triplicate to investigate changes in permeability in an effluent irrigated soil. The columns used for all experiments had a cross sectional area of 18.85 cm$^2$ and a length of 20 cm (only 10 cm of this length was packed with soil). Therefore, the volume of soil contained in the column was 10 cm x 18.85 cm$^2$ = 188.5 cm$^3$.

In this set of experiments, three columns (A, B and C) were packed as for the previous experiment. This was repeated to provide three identical columns of similar bulk density values (see Figure 4.8).

Once the columns were created, their contents were pre-wetted with 100 mL of rainwater under falling head conditions. After the rainwater had permeated through the soil, the first sample of permeant was taken. Eight solution additions were then made to each of the columns: 100 mL of rainwater, 250 mL of STE, 250 mL of STE, 100 mL of rainwater, 250 mL of STE, 50 mL of rainwater, 250 mL of STE and 100 mL of rainwater. The rainfall and STE had initial SAR values of 0.5 and 5.2 respectively. Each addition was applied when the previous addition had infiltrated into the soil. Actual rainwater and secondary treated effluent from the Branxton WWTW were used in this experiment.

Rainwater additions permeated under falling head conditions. For effluent additions, the first 150 mL of effluent permeated under constant head conditions and then under falling head conditions for the final 100 mL. The volumes 100 mL
(rainwater) and 250 mL (effluent) were chosen in an attempt to magnify any possible change in soil structure, due to the extreme change in SAR and electrolyte concentration between the two. One hundred millilitre of rainwater is equivalent to a 53 mm event, while 250 mL of effluent is equivalent to an irrigation event of 133 mm.

Rainwater was applied intermittently to mimic precipitation events that would provide conditions where the applied SAR was less than $C_{TU}$. ESP (soils) and SAR (solutions) calculations were performed using cation data from the output samples. The permeant was collected on a weight-over-time basis and analysed for pH, EC, turbidity and cations ($Ca^{2+}, Mg^{2+}, Na^+, K^+$) using to methods described in Section 4.3.1.

The effluent from the first time-based series (0 - 10 cm) was eluted through the next depth increment (10 - 20 cm) and collected, weighed and analysed in a similar fashion. All 10 – 20 cm additions were performed under falling head conditions. In addition, turbidity (NTU) was obtained from each sample to measure the degree of clay translocation out of the column. After the cycle of desired effluent loads had been
achieved, the soils were re-analysed for exchangeable cations and the differences recorded.

Saturated permeability ($K_{sat}$ – mm/hr) was determined from a calculation based on the volume of permeant application. The surface area (SA) of the column was obtained (0.00189 m$^2$), and using 1mm = 1L/m$^2$ and the time taken for each permeant sample to be collected, the following equation was used to determine $K_s$ in mm/hr:

$$K_{sat} = \left( \frac{\text{Eluted sample volume (L)}}{\text{Column area (m}^2\text{)}} \right) / \text{[Sample collection time (h)]}$$  \hspace{1cm} (Equation 20)

To determine whether advection and/or hydrodynamic dispersion played the dominant role in the transport of solutes through the profile of the receiving STE, solute breakthrough curves ($C/C_o$ versus time) were obtained. For example, when the column output water is of concentration ($C$), the sodium concentration in the output solution will be less than that of the effluent ($C_o$) initially applied. Figure 4.9 shows a solute breakthrough curve.

![Figure 4.9: Solute breakthrough curve ($C/C_o$) (after Hudak, 1999).](image-url)
When effluent of sodium concentration, $C_0$, is added to the column, the sodium concentration in the output solution (C) will increase over time (Figure 4.9). As more sodium reaches the outlet, C gradually increases to $C_0$. Had the solute been transported only by advection, C would rise suddenly rather than gradually to $C_0$ (Hudak, 1999). The solute “breaks through” when $C/C_0$ equals 0.5 (Shackelford, 1994).

4.4 Summary

The methods described in this chapter formed the basis for generating the data to investigate sodium flux with respect to daily and cumulative irrigation surplus/deficit, volumetric soil moisture, soil ESP, volume-weighted average SAR of effluent/rainfall and changes in groundwater depth. The relationship between the parameters measured and their rates of change over time provide insight into the temporal sodium flux of woodlot soils receiving STE.
Chapter 5: RESULTS

This chapter presents results from monitored field and laboratory parameters, column leaching experiments and soil moisture retention curves. Results are used to show how soil sodium may vary over time and also to highlight the woodlot soil’s potential response to the volume-weighted SAR of waters applied. In addition, the soil ESP/effluent SAR continuum for micro-aggregate/soil pore stability is conceptualised to show the dynamic variation in soil permeability under effluent irrigation.

5.1 Introduction

This chapter presents the results of this study in the order shown in Chapter 4. Section 5.2 presents results from irrigation scheduling based on several water balance components (Aim 1) and Section 5.3 presents results relating to the temporal sodium flux (Aim 2). Results from soil moisture characteristic curve determinations and column-leaching experiments are also presented in Section 5.3 and provide an insight for discussion with respect to the implications for irrigation scheduling and soil management (Aim 3). Section 5.4 describes the relationship between micro-aggregate/soil pore stability, soil ESP and effluent SAR, while Section 5.5 identifies dominant clay mineralogy in the studied soils by means of X-ray Diffraction (XRD) techniques. Section 5.6 summarises the most salient points emerging from the results presented.

All raw water balance data are contained in Appendix 1 with class A pan data in Appendix 2. Raw soil moisture data are shown in Appendix 3 with average soil moisture data in Appendix 4. All raw soil chemistry data are shown in full in Appendix 5. Groundwater depth data are shown in Appendix 6 with all chemical analyses for groundwater and effluent in Appendix 7. Raw column-leaching data for the 0 – 10 cm depth of soil and the 10 – 20 cm series are shown in Appendices 8 and 9 respectively.
5.2 Results of field measurements

5.2.1 Water balance components

The irrigation scheduling approach used in this study was based on the WATSKED model (Myers et al., 1999). WATSKED is an irrigation scheduling model that operates on a weekly time-step. The WATSKED model provided the pan coefficient values used in this study, sourced from Zone 6 PET data (Myers et al., 1999). The model itself was not adhered to directly in the scheduling of effluent, as the effluent volumes required to satisfy irrigation scheduling by weekly time-steps, could not be achieved at Branxton due to prolonged surface ponding. Using WATSKED irrigation recommendations would have resulted in excessive surface ponding and/or runoff at many times during January 2002 to October 2003.

As a result, the calculated cumulative irrigation surplus/deficit at a daily time-step was used as a surrogate for determining irrigation scheduling, based on volumetric soil moisture at the time of irrigation. The cumulative irrigation surplus/deficit has been used to summarise and describe the water balance at the study site between January 2002 and October 2003, as well as showing soil wetting and drying trends. Before discussion of water balance components, Figure 5.1 shows the comparison of cumulative irrigation surplus/deficit and WATSKED results.

Most irrigation scheduling volumes given by WATSKED were found to be excessive at the Branxton site, due to the fact that the model assumes steady-state conditions and uniform permeability through the root zone (Myers et al., 1999). However, WATSKED models soil water using a sprinkler irrigation efficiency value of 0.7 and it would be expected that WATSKED values would be greater in magnitude, as this relates to a 30% decrease in the actual value of Qe. Also, the use of crop factors much higher than 1.0 will increase the depth of effluent to be applied for a given day.
Figure 5.1: Comparison of cumulative irrigation surplus/deficit (ΔCI) and WATSKED trends (r² = 0.92), highlighting how WATSKED would have over-estimated irrigation demand at Branxton.
5.2.2 Applied effluent (Qe) and Precipitation (Qp)

The depth of effluent and rainfall that the woodlot soil received on a daily basis, between January 2002 and October 2003 is shown in Figure 5.2 and summarised in Table 5.1.

Figure 5.2: Depth of effluent and rainfall applied (January 2002 – October 2003)
Many assumptions of the water balance are made in effluent irrigation modelling, particularly with respect to soil structure. The assumed soil response is based on a steady-state system, which natural systems rarely obtain. Therefore, since soil texture and structure determines the ability of the soil to transmit water, results from water balance components are discussed in terms of the impact of varying soil ESP and volume-weighted SAR on soil structure and adaptive management of the receiving soil.

5.2.3 Potential Evapotranspiration (PET) and cumulative irrigation surplus/deficit ($\Delta CI$)

Discussion of this section will focus on PET and cumulative irrigation surplus/deficit, as these are the most relevant to achieving Aim 1 of this thesis. Transpiration and evaporation were discussed in Chapter 4 and have been incorporated into PET and the calculation of the daily irrigation deficit/surplus. The water balance used to determine the daily irrigation surplus/deficit ($\Delta I$) was as follows (Equation 11 from Chapter 2):

$$\Delta I = [PET + R + D - ((Q_p + Q_e) - IL)]$$

(Equation 11)

where $\Delta I =$ daily soil moisture deficit/surplus, $PET =$ potential evapotranspiration, $R =$ water lost as surface runoff, $D =$ water lost as percolation, $Q_p =$ precipitation, $Q_e =$ applied effluent and $IL =$ interception loss. All parameters are in millimetres (mm). Both $R$ and $D$ were assumed to equal zero, unless otherwise inferred by observation and/or fluctuating groundwater levels (discussed in Section 5.2.4). When groundwater levels were increasing, deep drainage was assumed to have occurred. Satisfying PET and producing zero surface runoff were considered optimum irrigation conditions. If the cumulative irrigation surplus/deficit and volumetric soil moisture are also increasing under optimum irrigation conditions, then leaching of sodium was assumed to have occurred.

Values for interception loss (IL) were incorporated to include a 9 % IL for summer months (September – March) and 6 % for winter months (April – August) and were discussed in Section 4.2.4. Table 5.1 shows the $K_p$ used to determine PET from evaporation data (Class A pan). $K_p$ was based on data from WATSKED, namely from
the Zone 6 - Coffs Harbour dataset in Myers et al. (1999).

Table 5.1: Pan co-efficients ($K_p$) used showing assumed variation in transpiration for each month

<table>
<thead>
<tr>
<th>MONTH</th>
<th>$K_p$</th>
<th>MONTH</th>
<th>$K_p$</th>
</tr>
</thead>
<tbody>
<tr>
<td>January</td>
<td>1.43</td>
<td>July</td>
<td>1.09</td>
</tr>
<tr>
<td>February</td>
<td>1.43</td>
<td>August</td>
<td>1.12</td>
</tr>
<tr>
<td>March</td>
<td>1.29</td>
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<td>1.13</td>
</tr>
<tr>
<td>April</td>
<td>1.15</td>
<td>October</td>
<td>1.17</td>
</tr>
<tr>
<td>May</td>
<td>1.13</td>
<td>November</td>
<td>1.23</td>
</tr>
<tr>
<td>June</td>
<td>1.11</td>
<td>December</td>
<td>1.39</td>
</tr>
</tbody>
</table>

Figure 5.3 shows PET between January 2002 and October 2003. PET is greater through the spring/summer months (September – February), although many above average PET values did occur in the winter months (June – August), particularly in 2002.

![Daily Average PET](image)

Figure 5.3 PET between January 2002 and October 2003
It is interesting to note that even though winter months had higher $K_p$ values, there is still a distinctive increase in PET for summer months versus winter months. The variation observed highlights the trees’ preference for water uptake during the summer months. PET ranged from 1 mm/day to approximately 18 mm/day with a daily average of 4.6 mm/day for January 2002 – October 2003. Total PET for the study period was approximately 3350 mm. Table 5.1 shows the pan co-efficients used for each month and for both years (2002 and 2003).

Figure 5.4 shows the daily irrigation surplus/deficit ($\Delta I$), based on PET, precipitation and applied effluent between January 2002 and October 2003. The majority of days ranged between zero and -20 mm (deficit), although many surplus events occurred over the same period. These included several “spikes” of approximately 80 mm in magnitude, in February, June and December 2002.

Figure 5.5 shows the cumulative irrigation surplus/deficit between January 2002 and October 2003 and it is this graph, which is used during discussion of trends in monitored parameters in Chapter 6.
Figure 5.5: Cumulative irrigation surplus/deficit from January 2002 and October 2003 highlighting wetting and drying trends.
For example, if 15 mm of water was lost (deficit) on a given day, then the daily \( \Delta I \) would be -15 mm; therefore \( \Delta CI = -15 \) mm. If no effluent/rainfall was applied the following day and another 15 mm was lost, then the daily \( \Delta I \) would still be -15 mm, although the cumulative irrigation surplus/deficit (\( \Delta CI \)) would equal -30 mm. Therefore, the cumulative irrigation surplus/deficit represents the pattern for lost irrigation opportunities and excessive irrigation events between January 2002 and October 2003.

The zero line represents the site when soil saturation and excessive surface ponding was observed (5\(^{th}\) of February 2002). On two occasions the saturation line was exceeded, throughout February 2002 and the period from May to August 2002. From Figure 5.5, surplus periods include the first week of February 2002, April – June 2002, December 2002 – mid January 2003 and mid February – June 2003. Deficit periods include mid February – April 2002, June – December 2002, mid January – mid February 2003 and June – October 2003. These periods are numbered 1 to 7 in Figure 5.5 and represent major surplus and deficit soil water trends.

Figure 5.5 shows that the woodlot was under irrigated with respect to the daily irrigation deficit/surplus between June and December 2002, as very few days registered a surplus value. From February 2003 until June 2003, daily hydraulic loads were more evenly distributed, which is apparent from the increased frequency of surplus events. From late June until late September 2003, there is a decrease in the frequency of surplus irrigation/rainfall events. All of the above represent different wetting and drying conditions that would influence sodium movement in the soil profile.

5.2.4 Groundwater

Groundwater samples were collected on a monthly basis, as described in Section 4.2.7, and from sites shown in Figure 4.5. The figures presented during this chapter are summaries of selected parameters, with the full data set included in Appendices 6 and 7. Depth to groundwater was determined on a weekly basis. Figure 5.6 shows the variation in the depth to groundwater from the surface between June 2002 and December 2003 for four sites.
The surface elevation of all sites is between 36 and 37 m AHD. The lowest surface elevation is GW2 (36 m), followed by GW3 (36.1 m), GW1 (36.4 m) and GW5 (36.6 m). During May 2002, groundwater depth below the surface at GW1 was approximately 1.8 m, GW2 at 1.1 m, GW3 at 2.6 m and GW5 at 1.0 m, until June 2002. Due to excessive rainfall in June 2002, the depth to groundwater at GW2 decreased from approximately 1.1 m to 0.6 m, GW5 decreased from approximately 1.0 m to 0.8 m, GW1 decreased from 1.8 m to 1.6 m, and GW3 decreased from approximately 2.6 m to 2.5 m.

Figure 5.6: Depth to groundwater monitored on a weekly basis (April 2002 – December 2003) showing the wetting and drying trends (sub-periods) identified in Figure 5.5

All increases at this time coincide with rainfall experienced in late May/early June 2002 (Figure 5.2). GW2 appeared to be the most responsive to excessive rainfall/effluent loading as it experienced the greatest increase after the late May/early June 2002 event and was nearest the surface for most of the study period. However,
from May 2003 both GW2 and GW5 have similar groundwater depths ranging from 1.3 to 1.4 m. In late October 2003, groundwater levels at GW2 rose nearer the surface in response to October effluent irrigation and rainfall ($Qe + Qp = 172.8$ mm).

Groundwater hydraulic conductivity ($K_s$) was determined by methods discussed in Section 4.2.7. The purpose of determining groundwater $K_s$ was to gain insight into the rate at which groundwater flows through the site in the phreatic zone. The slug-injection test provided a one-off result, as continual monitoring of $K_s$ would have disrupted other parameters that were considered more relevant to this thesis. For example, persistent slug injection would cause changes to groundwater depth and chemistry and would have masked any significant change in groundwater property comparisons. Data obtained from the slug-injection showed $K_s$ to be approximately 0.90 mm/hr (Table 5.2). Figure 5.7 shows the direction of groundwater flow through Area 3 at the Branxton WWTW.

The purpose of determining groundwater direction was to investigate if specific boreholes were more responsive to hydraulic loading than others, thus representing indicators for excessive irrigation scheduling. GW2 exhibits a greater response to hydraulic loading due to its proximity to the surface, and if it could be proven that GW2 lies at the point of subsurface convergence, then use of GW2 depth records can provide information on the extent of deep drainage. It appears that subsurface water would be directed through GW2. Since GW2 is topographically the lowest site in area 3, it follows that the depth of groundwater at GW2 provides an adequate monitoring parameter in assessing the potential for surface runoff in Area 3, from irrigation and/or rainfall.

### Table 5.2: Slug-injection data - summary

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<th>Finish: Friday 20 June, 2003: 9am</th>
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<td>$F$ was determined from Equation 14, $K$ by Equation 20.</td>
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</tbody>
</table>

<table>
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<th>Area</th>
<th>$F$</th>
<th>$H1$</th>
<th>$H2$</th>
<th>$t1$</th>
<th>$t2$</th>
<th>$L$</th>
<th>$r$</th>
<th>$K_s$</th>
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<td>2000</td>
<td>0</td>
<td>336</td>
<td>1000</td>
<td>25</td>
<td><strong>0.90</strong></td>
</tr>
</tbody>
</table>
5.2.5 Soil survey results – site selection and soil sampling

Individual soil profile descriptions for Area 3 were obtained from initial site assessment and these are presented during this chapter in conjunction with soil moisture data. A conceptual description and classification of the study site soils is shown on the following page (Figure 5.8).

Figure 5.8 describes the soil sites in terms of horizon boundary and observed texture. The six sites selected for soil monitoring/analyses include S1, S5, S8, S9, S10, and S14 (Figure 4.5, Chapter 4). Sites S2, S4, S6 and S13 are described in terms of soil texture with depth and volumetric soil moisture (θ) only, as no soil samples were taken at these sites after initial sampling on the 18/12/01 and 20/2/02. S3, S7 and S11 were unsuitable over the study period, due to access tube damage and/or leakage, which compromised data quality. S9 and S14 exist outside the irrigation area and are used as background non-irrigated sites.
Figure 5.8: Observed horizon boundaries and textures of sampled sites at Branxton WWTW
Apart from a small detritus layer (<0.5 cm) on the surface, the soil texture at S1, S2, S3, S8 and a non-irrigated site S14 remained relatively unchanged with depth (see Figure 5.8). That is, no horizon boundary was found that was anticipated to impede permeability to the sampled depth of 80 cm. The soil profile at S4, S5, S6, S7, S10, S11, S12, S13 and a non-irrigated S9 had sandy clay at depths ranging from 30 cm to 60 cm.

Results for each site are presented separately and show the time series for volumetric soil moisture, measured on a weekly basis. In addition, monitored $\theta$ values from all irrigated sites in the woodlot were averaged. The average $\theta$ results are meant to be representative of the entire site, showing the dominant trends in $\theta$ over time.

The soil sample sites selected represent the wettest and driest sites in the woodlot. Furthermore, the non-irrigated sites S9 and S14 were selected due to their similarity in soil profile and texture, as compared to sites within the woodlot. For example, irrigated sites with loamy sand to the sample depth of 80 cm include S1 and S8, with S8 being much wetter than S1. The irrigated sites consisting of both loamy sand (A horizon) and sandy clay (B horizon) included S5 and S10 although $\theta$ values were similar with respect to soil saturation, particularly at depths greater than 10 cm. The non-irrigated sites S9 and S14 comprised of loamy sand/sandy clay and loamy sand respectively, to the sampled depth of 80 cm.

Soil extraction plate methods and column-leaching experiments were performed on the representative soil sample collected on the 18/12/01 and 20/2/02. In reporting site-specific $\theta$, the following sections refer to previously presented figures in this chapter. Unfortunately, $\theta$ data for parts of April/May 2002 were erroneously deleted from the archive file and also over-written in the Gopher data logger for S2, S4, S6, S13 and the non-irrigated site S9 and no data exists for this two month period.
5.2.6 Site-specific volumetric soil moisture (θ)

Site 1 – S1

Figure 5.9 shows that θ increased in February 2002 at all recorded depths, which corresponded to rainfall events during the first week of February 2002 (see Figure 5.2). The θ at this time was 29.6 % at 10 cm, 15.2 % at 40 cm, 21.4 % at 60 cm and 29.3 % at 80 cm and were the highest values obtained over the study period. For each recorded depth, θ at 10 cm ranged from 4.7 % to 29.6 %; 2.9 % to 15.2 % at 40 cm; 4.0 % to 21.4 % at 60 cm and 5.1 % to 29.3 % at 80 cm from January 2002 – October 2003.

Irrigation in April 2002 (refer Figure 5.2) increased θ at this time. θ decreased at depths of < 40 cm until December 2002, despite prolonged rainfall in late May/early June 2002 (refer Figure 5.2). From August 2002 – December 2002, θ was often higher at a depth 80 cm than θ at either 40 or 60 cm depth. At 10 cm, the θ was relatively higher than all other depths from May 2002 – December 2002, reflecting the consistent
hydraulic loading (refer Figure 5.2), although negligible volumes appear to reach lower depths.

During December 2002, $\theta$ increased at all recorded depths in response to rainfall (refer Figure 5.2). By the end of January 2003, the highest $\theta$ was at 80 cm (20\%). The hydraulic load in March/April 2003 caused an increase in $\theta$ at all depths. From this point until October 2003, $\theta$ at 10 cm fluctuated while $\theta$ at 40 and 60 cm were low and stable. Volumetric soil moisture at 80 cm depth decreased from February 2003 – May 2003, then increased in July 2003, before returning to May 2003 values until October 2003.

Generally, volumetric soil moisture was greater at the surface (10 cm) for most of the study period. This observation was common to all irrigated sites that comprised of loamy sand (S1, S2 and S8) to the sampled depth of 80 cm.

**Site 2 – S2**

Figure 5.10 shows that $\theta$ increased in February 2002 at all recorded depths, which corresponded to rainfall events during the first week of February 2002 (see Figure 5.2). $\theta$ at 10 cm ranged from 2.1 % to 22.4 %; 4.5 % to 17.2 % at 40 cm; 4.7 % to 17.4 % at 60 cm and 5.4 % to 21.6 % at 80 cm from January 2002 – October 2003.

Maximum $\theta$ values occurred in February 2002, June 2002, and December 2002. Minimum $\theta$ values occurred in November/December 2002. Irrigation in April 2002 (refer Figure 5.2) increased $\theta$ at this time, then $\theta$ decreased at depths of 40 cm, 60 cm and 80 cm until December 2002, despite prolonged rainfall in late May/early June 2002 (refer Figure 5.2). From August 2002 – December 2002, $\theta$ was often higher at 80 cm than at either 40 cm or 60 cm. At 10 cm, the $\theta$ was relatively higher than all other depths from May 2002 – December 2002, reflecting the consistent hydraulic loading (Figure 5.2), although limited volumes appear to reach lower depths.

During December 2002, Figure 5.10 shows that all depths recorded an increase in $\theta$ in response to rainfall (refer Figure 5.2). By the end of January 2003, the highest $\theta$ was at 80 cm (~7 \%). The hydraulic load in March/April 2003 caused an increase in $\theta$ at
all recorded depths. From this point until October 2003, $\theta$ at a depth of 10 cm fluctuated significantly while $\theta$ at 40 cm and 60 cm were low and stable. Volumetric soil moisture at 80 cm depth decreased from February 2003 – May 2003, then increasing in July 2003, before returning to May 2003 values until October 2003. Similarly to S1, $\theta$ at S2 is greater near the surface (10 cm) than at lower depths, particularly during 2003.

![Graph](image)

**Figure 5.10: Volumetric soil moisture ($\theta$) at S2 (January 2002 – October 2003)**

**Site 4 – S4**

The presence of the clay horizon is apparent in Figure 5.11. For example, while $\theta$ at 10 cm depth did not exceed 33 %, $\theta$ at other depths rarely fell below 33 %. Conversely, $\theta$ at 40 cm, 60 cm and 80 cm depth were approximately twice the value of sites previously discussed and it follows that this is indicative of the higher moisture holding capacity of clays. The response to hydraulic loads shows the same peaks at maximum effluent/rainfall (refer Figure 5.2) as S1, for example, in January 2002, June 2002 and December 2002. Generally, S4 was saturated at depths of less than 40 cm, indicating high storage of soil water at these depths. In contrast to sites without a clay
horizon to the sampled depth of 80 cm, S4 had lower $\theta$ at 10 cm than other depths, particularly from April 2002 onwards.

Site 5 – S5

S5 had a clay horizon at 30 cm (Figure 5.8). Figure 5.12 shows $\theta$ between January 2002 and October 2003 and the response of the soil to irrigation/rainfall events over time. Unfortunately, the access tube was crushed by a tractor during thinning operations and was not re-established after January 2003.

However, the presence of the clay horizon is apparent in Figure 5.12. For example, while $\theta$ for the 10 cm depth never exceeded 35 %, $\theta$ at greater depths rarely fell below 35 %. $\theta$ at 40 cm depth did not markedly decrease until September 2002. The response to hydraulic loads shows the same peaks at maximum effluent/rainfall (refer Figure 5.2) as other irrigated sites, for example, in January 2002, June 2002 and December 2002. S5 is similar to S4 in the fact that $\theta$ is lower at 10 cm and saturated at lower depths, which is in contrast to irrigated sites which show wetter (surface) to drier
(at depth) $\theta$ profiles (for example, S1, S2 and S8).

Figure 5.12: Volumetric soil moisture ($\theta$) at S5 (January 2002 – February 2003)

Site 6 – S6

S6 had a sandy clay horizon at 40 cm (Figure 5.8). Figure 5.13 shows weekly $\theta$ between January 2002 and October 2003 and the response of the soil to irrigation/rainfall events over time. The presence of the clay horizon is apparent in Figure 5.13. For example, while $\theta$ for the 10 cm depth never exceeded 38 %, $\theta$ at 40 cm never fell below 27 % and $\theta$ at 60 cm and 80 cm never fell below 35 %.

Volumetric soil moisture at a depth of 60 cm and 80 cm are approximately twice the value of sites without clay horizons and this is indicative of the higher moisture holding capacity of clays. However, while $\theta$ at 40 cm depth rarely falls below $\theta$ at 10 cm, $\theta$ at 40 cm depth also never exceeds $\theta$ at lower depths, showing the lower clay content at 40 cm than that at 60 cm or 80 cm. The response to hydraulic loading shows the same peaks at maximum effluent/rainfall (refer Figure 5.2) as other sites, for example, in January 2002, June 2002 and December 2002.
Over time, the access tube at Site 7 (S7) developed a leak that compromised $\theta$ values and as a result, was neglected for the purposes of this study.

**Site 8 – S8**

Figure 5.14 shows that $\theta$ increased in February 2002 at all recorded depths, which corresponded to rainfall events during the first week of February 2002 (see Figure 5.2). Volumetric soil moisture at 10 cm and 40 cm were greater than $\theta$ at 60 cm and 80 cm from February 2002 until November 2002, where $\theta$ at 10 cm and some $\theta$ values at 40 cm are lower than $\theta$ at 60 cm an 80 cm. This would indicate that S8 was saturated or near-saturated for most of this time.

The 10 cm depth had the highest $\theta$ from February 2002 to November 2002, ranging from 20 % to 39 %. $\theta$ values at 40 cm, 60 cm and 80 cm ranged from approximately 15 % to 37 %, 28 % to 32 % and 25 % to 33 % respectively. After December 2002 rainfall, $\theta$ increased at all sites. Volumetric soil moisture at 10 cm fluctuated significantly between February 2003 and October 2003, ranging from 11 % to
This fluctuation was due to irrigation scheduling and rainfall. Volumetric soil moisture values at 40 cm, 60 cm and 80 cm ranged from approximately 23 % to 32 %, 24 % to 34 % and 23 % to 36 % respectively.

![Volumetric soil moisture (θ) at S8 (January 2002 – October 2003)](image)

**Figure 5.14: Volumetric soil moisture (θ) at S8 (January 2002 – October 2003)**

**Site 9 – S9 (non-irrigated site – with sandy clay B horizon)**

Figure 5.15 shows θ between January 2002 and October 2003 and the response of the soil to rainfall over time. The presence of the clay horizon is apparent in Figure 5.15. For example, while θ for the less than 40 cm depth never exceeded 28 %, θ at > 40 cm rarely fell below 35 %. θ values at > 80 cm depth are approximately twice the value of θ at less than 40 cm. Sandy clay was present at a depth greater than 40 cm.

The response to hydraulic loads shows the same peaks at maximum rainfall (refer Figure 5.2) as other sites, for example, in January 2002, June 2002 and December 2002. In contrast to effluent irrigated sites, θ at 10 cm fluctuates less from week to week with θ values ranging from 1 to 20 %. Volumetric soil moisture values at 40 cm, 60 cm and 80 cm depths ranged from approximately 2 % to 32 %, 32 % to 58 % and 30 % to
Volumetric soil moisture at 60 cm and 80 cm showed saturated conditions for most of the study period, while the surface 10 cm was consistently drier than lower depths.

Site 10 – S10

Monitoring of S10 began in March 2002, as it was in an area of the woodlot where prolonged surface ponding was observed in February 2002. Approximately four weeks was required for the surface water to infiltrate before installation of the soil moisture access tube could occur. The soil at S10 had an abrupt B-horizon at 20 cm (Figure 5.8) and was observed to be near-saturated, particularly at depths greater than 40 cm, for most of the sampling period. Figure 5.16 shows θ between January 2002 and October 2003 and the response of the soil to irrigation/rainfall over time.

The presence of the clay horizon is less obvious in Figure 5.16, than it is at other sites. For example, θ at > 40 cm depths rarely fell below the corresponding θ value at 10 cm. However, θ values at 10 cm depth exceed 30% on several occasions throughout
the study period in response to effluent irrigation and rainfall. The response to hydraulic loads shows less distinctive peaks at maximum rainfall (refer Figure 5.2) as other sites, for example, in June 2002 and December 2002.

At the surface (10 cm), $\theta$ is highly variable from March 2003 to October 2003, with $\theta$ ranging from 6 % to 35 %. $\theta$ values at 40 cm, 60 cm and 80 cm depths ranged from approximately 27 % to 34 %, 22 to 34 % and 32 to 37 % respectively. In addition, the $\theta$ values at depths of less than 40 cm are greater than those at 60 cm, possibly showing a perched water table. Interestingly, $\theta$ at depth at S9 (non-irrigated) had higher or similar $\theta$ to S10 (irrigated site) for much of the study period. Site 11 (S11) was destroyed by cattle from the adjacent property and was never replaced.

**Site 12 – S12**

Figure 5.17 shows $\theta$ values between January 2002 and October 2003 and the response of the soil to irrigation/rainfall over time. The response to hydraulic loading showed less distinctive peaks at maximum rainfall (refer Figure 5.2) as other sites, for
example, in January 2002, June 2002 and December 2002. This was due to clay being present at 30 cm and impeding downward water movement.

![Diagram of soil moisture values](image)

**Figure 5.17: Volumetric soil moisture (\(\theta\)) at S12 (March 2002 – October 2003)**

The 10 cm depth showed the greatest response to irrigation/rainfall from February 2003 to October 2003, with \(\theta\) values ranging from 14 % to 32 %. \(\theta\) values at 40 cm, 60 cm and 80 cm depths ranged from approximately 17 % to 34 %, 22 % to 35 % and 30 % to 58 % respectively. Between November 2002 and April 2003, \(\theta\) at 80 cm depth decreased significantly from 55 % to 30 %, before returning to pre-November 2002 values (57 %). Clay at 80 cm underwent rapid drying during late October/early November 2002, with a similarly rapid wetting response in late April 2003.

**Site 13 – S13**

Monitoring of S13 began in March 2002 because S13 was in an area of the woodlot where prolonged surface ponding was observed in February 2002. As a result, approximately four weeks was required for the surface water to infiltrate before
monitoring could begin. The soil at S13 had an abrupt B-horizon at 40 cm, similar to that of S6 (Figure 5.8). Figure 5.18 shows $\theta$ between January 2002 and October 2003 and the changes in $\theta$ of the soil to irrigation/rainfall over time.

![Figure 5.18: Volumetric soil moisture (\(\theta\)) at S13 (March 2002 – October 2003)](image)

The presence of the clay horizon is apparent in Figure 5.18. For example, $\theta$ at depths less than 40 cm rarely fell below 40 % (saturated). However, $\theta$ values at less than 40 cm depth never exceeded 37 % throughout the study period. The response to hydraulic loading showed less distinctive peaks at maximum rainfall (refer Figure 5.2) as other sites, for example, in January 2002, June 2002 and December 2002. $\theta$ at 10 cm depth fluctuated from March 2002 to October 2003, with $\theta$ values ranging from 14 to 35 %. $\theta$ values at 40 cm, 60 cm and 80 cm depths ranged from approximately 27 % to 35 %, 38 % to 55 % and 40 % to 55 % respectively for the same period.
Site 14 – S14 (non-irrigated site – without sandy clay B horizon)

S14 existed outside the woodlot (Figure 3.4) and was selected (mid-May 2002) as the non-irrigated site for other sites with no clay B horizon present in the sampled depth of 80 cm. Figure 5.19 shows $\theta$ between January 2002 and October 2003 and the response of the soil to rainfall over time. The response to hydraulic loading shows the same peaks at maximum rainfall (refer Figure 5.2) as other sites, for example, in June 2002, December 2002, late February 2003.

![Figure 5.19: Volumetric soil moisture ($\theta$) at S14 (March 2002 – October 2003)](image)

In contrast to effluent irrigated sites, $\theta$ at 10 cm lower and more variable from week to week, responding rapidly to any rainfall. Volumetric soil moisture values at 10 cm ranged from 1 to 18 %. Volumetric soil moisture values at 40 cm, 60 cm and 80 cm depths ranged from approximately 10 % to 20 %, 8 % to 18 % and 11 % to 17 % respectively. Volumetric soil moisture at all sites is generally lower than effluent irrigated sites and is due to the lower hydraulic load received. Comparison between non-irrigated and irrigated sites are made in Section 5.3.5.
5.2.7 Average weekly $\theta$ of all sites within woodlot

Weekly volumetric soil moisture ($\theta$) was monitored to validate wetting/drying periods identified in $\Delta I$ and cumulative irrigation surplus/deficit (Section 5.2.3). As such, weekly $\theta$ results have been used to validate identified trends in cumulative irrigation surplus/deficit over time. Figure 5.20 shows $\theta$ values measured each week, for the 10, 40, 60 and 80 cm depths.

Average results were determined for use in conjunction with column-leaching experiments and soil moisture characteristic curves, both of which were determined using a representative sample obtained on the 18/12/01 and 20/2/02. Maximum, minimum, range and RSD of averaged results are shown in Appendix 4. The $\theta$ values presented are the average values for all weekly monitoring performed at sites discussed in Chapter 3, namely S1, S2, S4, S5, S6, S8, S10, S12 and S13. Average $\theta$ values for the non-irrigated sites S9 and S14 are not included here. They have been presented individually and will be discussed during Section 5.3.5.

![Average weekly volumetric soil moisture](image)

Figure 5.20: Weekly volumetric soil moisture – averaged from nine sites (Jan 2002 – Oct 2003).
Figure 5.20 shows that average $\theta$ values increased at all sites during February 2002 due to rainfall. Apart from a slight decrease in average $\theta$ values at all depths at the end of February 2002, $\theta$ values increased at all depths during March 2002. Between March 2002 and late May 2002, average $\theta$ values decreased, before increasing again at the beginning of June 2002.

From June 2002 until December 2002, all depths recorded a decrease in average $\theta$ values. Average $\theta$ values at 10 cm experienced the largest variation and decline in average $\theta$ values, particularly through October/November 2002. Average $\theta$ values at 40 cm, 60 cm and 80 cm rarely fell below 22 % for the same period and appear similar in their rate of decline. It is likely that the variability between textural horizons between sample sites has skewed the presented results however the general wetting and drying trends indicative of efficient irrigation are clearly shown.

During the first week of December 2002, average $\theta$ values at all depths increased to values similar to those in February 2002. By the first week of February 2003, all depths showed a decrease in average $\theta$, similar to those experienced in early December 2002. Average $\theta$ values at 40 cm, 60 cm and 80 cm showed an increase from February 2003 until May/June 2003, where average $\theta$ then remained relatively stable until decreasing in September/October 2003.

Average $\theta$ at 10 cm depth fluctuated the most throughout the monitoring period, with average $\theta$ values ranging from 12 % to 32 %. The average $\theta$ values at 40 cm depth ranged from 16 % to 33 %, 22 % to 38 % at 60 cm and 24 % to 40 % at 80 cm. Ultimately, it was trends in average $\theta$ values that validated the wetting/drying periods presented in Section 5.2.3 and seen in Figure 5.5. Figure 5.20 shows that average $\theta$ generally increased with an increase in depth. However, specific irrigated sites were observed to show the opposite, that is, volumetric soil moisture decreased with depth. Irrigated sites S1, S2 and S8 exhibited this trend for most of the study period as a result of effluent irrigation.
5.3 Results of laboratory measurements

5.3.1 Effluent and rainwater analyses

The methods of collection and analyses of effluent and rainfall were discussed in Section 4.3.1. Table 5.3 summarises monthly effluent results. Rainfall SAR was approximately 0.5 and was determined from a “one-off” analysis (last row of Table 5.3). Due to the low concentrations of electrolytes in rainwater (Ca$^{2+}$, Mg$^{2+}$, Na$^{+}$ and K$^{+}$), the effect of small variations in the SAR of rainfall was considered negligible. Rainfall SAR was then assumed to be constant at the site throughout the study period. Figure 5.21 shows the daily volume-weighted average SAR of the applied irrigated waters and rainfall from June 2002 – October 2003.

**Table 5.3: Effluent/rainfall chemical analyses from monthly sampling**

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<td>27-Nov-02</td>
<td>STE</td>
<td>8.04</td>
<td>1120</td>
<td>1.64</td>
<td>0.50</td>
</tr>
<tr>
<td>26-Dec-02</td>
<td>STE</td>
<td>6.91</td>
<td>830</td>
<td>1.47</td>
<td>0.49</td>
</tr>
<tr>
<td>29-Jan-03</td>
<td>STE</td>
<td>7.71</td>
<td>845</td>
<td>1.39</td>
<td>0.48</td>
</tr>
<tr>
<td>27-Feb-03</td>
<td>STE</td>
<td>6.52</td>
<td>880</td>
<td>1.41</td>
<td>0.60</td>
</tr>
<tr>
<td>10-Apr-03</td>
<td>STE</td>
<td>6.48</td>
<td>865</td>
<td>0.79</td>
<td>0.45</td>
</tr>
<tr>
<td>25-Apr-03</td>
<td>STE</td>
<td>6.16</td>
<td>840</td>
<td>1.39</td>
<td>0.48</td>
</tr>
<tr>
<td>31-May-03</td>
<td>STE</td>
<td>6.33</td>
<td>815</td>
<td>1.33</td>
<td>0.41</td>
</tr>
<tr>
<td>25-Jun-03</td>
<td>STE</td>
<td>6.25</td>
<td>870</td>
<td>1.43</td>
<td>0.42</td>
</tr>
<tr>
<td>31-Jul-03</td>
<td>STE</td>
<td>6.18</td>
<td>920</td>
<td>1.58</td>
<td>0.52</td>
</tr>
<tr>
<td>28-Aug-03</td>
<td>STE</td>
<td>6.28</td>
<td>960</td>
<td>1.46</td>
<td>0.46</td>
</tr>
<tr>
<td>26-Sep-03</td>
<td>STE</td>
<td>6.47</td>
<td>680</td>
<td>1.56</td>
<td>0.64</td>
</tr>
<tr>
<td>31-Oct-03</td>
<td>STE</td>
<td>6.78</td>
<td>690</td>
<td>1.37</td>
<td>0.46</td>
</tr>
<tr>
<td>8-Feb-02</td>
<td>Rainwater</td>
<td>5.74</td>
<td>100</td>
<td>0.24</td>
<td>0.04</td>
</tr>
</tbody>
</table>
SAR values obtained from monthly effluent analyses were assumed constant for that month and were used in the mixing model for daily calculations. Rainfall SAR was assumed to be constant. The volume-weighted average SAR of applied effluent and rainfall was determined using the mixing model equation from Chapter 4 (Equation 19). STE and rainwater SAR values were determined from methods discussed in Section 4.3.1. The SAR of the applied STE ranged from 3.2 to 5.9 from monthly analyses. In Figure 5.21, each column represents the daily volume-weighted average SAR of the applied STE and/or rainfall. No column means zero effluent irrigation or rainfall occurred at those times.

**5.3.2 Groundwater analyses**

Figures 5.22 A – C show monthly groundwater pH, EC (µS/cm), and SAR respectively, between June 2002 and November 2003. All groundwater sites had different pH values on each sampling occasion. GW1 always had the lowest pH, which ranged from 4.4 to 5.9. GW2, GW3 and GW5 had pH ranges of 6.4 – 7.0, 6.5 – 7.1 and 7.1 – 7.3, respectively. At all sites, pH values show little variation throughout the
Figure 5.22 (A – C): Monthly groundwater summary analyses: (A) pH; (B) EC (µS/cm); (C) SAR (June 2002 – November 2003)
monitoring period, although the pH variation between sites is repeated on every sampling occasion.

GW2 consistently had the lowest of all EC values while GW3 consistently had the highest of all EC values. However, the EC range for all sites between June 2002 and November 2003 is large. The August – September 2002 and September – November 2003 periods exhibit the largest increase in EC, for all sites, of approximately 4000 and 9000 µS/cm respectively.

Compared to June 2002 sampling/analysis, significantly higher EC values were recorded for September, October and November 2002, and October – November 2003. Other periods remained relatively stable. EC for GW1 ranged from approximately 1900 – 12000 µS/cm, GW2 from approximately 1950 – 9400 µS/cm, GW3 from approximately 2700 – 13400 µS/cm and GW5 from approximately 2300 – 9200 µS/cm, between June 2002 and November 2003.

Groundwater SAR increased for all sites between June 2002 and September 2002. From this point onward, the SAR for all sites remained relatively stable, although GW1 and GW2 had consistently lower SAR values than GW3 and GW5. September 2003 is an anomaly in the dataset due to the extreme SAR. The SAR for GW1, GW2 and GW5 near or exceed 100, whilst the SAR of GW3 was approximately 60. These values are significantly different to previously monitored SAR values for these groundwater sites and will be discussed in Chapter 6.

Figure 5.22 D shows groundwaters are predominantly of a sodium type. Although not measured in this study, analyses of several groundwater samples in the past have shown chloride to be the major anion present (Dennis and Sinclair, 1994). Anions were not measured because the analytical instruments, costs and time were limited. The predominance of sodium in the groundwater at the Branxton site is typical of other groundwaters in the region, as are the large spatial variations in EC observed throughout the Hunter Valley (Beale et al., 2000). Table 5.4 shows the variation in EC of ground waters sourced from Permian geology in the Hunter Valley and “n” equals the number of bores monitored.
Figure 5.22 D: Ternary plot for measured groundwater cations (June 2002 – October 2003) 
(Ca$^{2+}$, Na$^+$, Mg$^{2+}$)

Table 5.4: Salinity of groundwater sourced from the Permian 
(extracted from Beale et al., 2000)

<table>
<thead>
<tr>
<th>Geology</th>
<th>Location</th>
<th>Electrical conductivity (dS/m)</th>
<th>Min.</th>
<th>Max.</th>
<th>Average</th>
<th>StDev</th>
<th>n</th>
</tr>
</thead>
<tbody>
<tr>
<td>Late Permian</td>
<td>Central</td>
<td></td>
<td>0.380</td>
<td>25.8</td>
<td>3.649</td>
<td>3.716</td>
<td>74</td>
</tr>
<tr>
<td></td>
<td>South-east</td>
<td></td>
<td>0.169</td>
<td>5.7</td>
<td>1.542</td>
<td>1.540</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>West</td>
<td></td>
<td>0.226</td>
<td>7.6</td>
<td>1.579</td>
<td>1.320</td>
<td>88</td>
</tr>
<tr>
<td></td>
<td>All</td>
<td></td>
<td>0.169</td>
<td>25.8</td>
<td>2.393</td>
<td>2.753</td>
<td>187</td>
</tr>
<tr>
<td>Early Permian</td>
<td>Central</td>
<td></td>
<td>0.630</td>
<td>9.5</td>
<td>3.387</td>
<td>2.874</td>
<td>11</td>
</tr>
<tr>
<td></td>
<td>South-east</td>
<td></td>
<td>0.373</td>
<td>9.3</td>
<td>2.280</td>
<td>2.222</td>
<td>22</td>
</tr>
<tr>
<td></td>
<td>All</td>
<td></td>
<td>0.373</td>
<td>9.5</td>
<td>2.649</td>
<td>2.471</td>
<td>33</td>
</tr>
</tbody>
</table>

5.3.3 Soil analyses - temporal soil ESP, ΔESP, change in sodium (cmol(+) / kg) at specific sites

This section presents the results of monitoring soil sodium every two months from February 2002 to October 2003. Results for soil ESP, change in ESP (ΔESP) and change in sodium (cmol(+) / kg) between sample times are shown. The approach taken in this study effectively provided 11 two-monthly monitoring periods between February 2002 and October 2003. In this section, the purpose of describing the subtle changes in
soil sodium is to highlight just how dynamic soil sodium was during this study and relate the observed trends to water balance components.

The consequences for not including exchange acidity (EA) in the calculation of cation exchange capacity are minor. Figure 5.23A shows the relationship between pH and EA for the selected soils and Figure 5.23B highlights the minor variation from not including EA in CEC calculations and with CEC calculated with inclusion of EA. As a result, the influence on soil ESP calculations would also be minor and would not seriously detract from the robustness of results as presented in this thesis.

Figure 5.23: (A) pH versus Exchange Acidity (EA); and (B) ESP (no EA) versus ESP (with EA)
The variability at a given site was determined at S8 by duplicate sampling, and is discussed during this section. Duplicate sampling was performed to investigate whether the change in soil ESP at the time of sampling was indicative of the soil ESP change from period to period, or whether soil ESP has a natural at-a-site variance at the time of sampling. To provide a datum for determining soil ESP from period to period, soil ESP values for February were designated zero to calculate subsequent increases/decreases in $\Delta$ESP and change in sodium (cmol(+)/kg) for April 2002 results and subsequent variations are relative to the previous sampling period.

Site 1 – S1

Figure 5.24 shows the ESP and change in ESP ($\Delta$ESP), between sampling periods, for S1 between February 2002 and October 2003. The $\Delta$ESP reflects the amount of sodium lost or gained from one monitoring period to another. ESP values at 10 cm ranged from approximately 11% to 26% for the entire monitoring period. At 20 cm, 40 cm, 60 cm and 80 cm, ESP values ranged from approximately 11% to 36%, 12% to 34%, 18% to 33% and 18% to 48% respectively.

For April 2002, the columns in Figure 5.24 represent the $\Delta$ESP for February 2002 to April 2002. All depths recorded an increase in ESP during this period. At 10 cm depth, $\Delta$ESP had risen approximately 3%, 9% at 20 cm, 6% at 40 cm, 1% at 60 cm and 12% at 80 cm. Soil ESP during the first monitoring period showed significant variation from February soil ESP results. Compared to April 2002 $\Delta$ESP values, all depths except 80 cm recorded a further increase in ESP in June 2002. At 10 cm depth, $\Delta$ESP had risen by approximately 4%, 3% at 20 cm, 12% at 40 cm and 16% at 60 cm. At 80 cm, $\Delta$ESP had decreased by approximately 2%. In addition, ESP generally increases with depth for most of the study period.
Figure 5.24: Soil ESP, ΔESP and change in sodium (cmol(+)/kg) at S1 (February 2002 – October 2003)
At all depths, further increases in soil ESP occurred for August 2002. At 10 cm depth, ΔESP had risen approximately 8 %, 14 % at 20 cm, 5 % at 40 cm, 2% at 60 cm and 21 % at 80 cm. Compared to August 2002, soil ESP for October 2002 decreased significantly. At 10 cm depth, ΔESP had fallen approximately 8 %, 16 % at 20 cm, 13 % at 40 cm, 12 % at 60 cm and 23 % at 80 cm.

Soil ESP during December 2002 decreased at ≤ 20 cm depths and increased at ≥ 20 cm depths. At 10 cm depth, ΔESP further decreased by approximately 4 % and 2 % at 20 cm. ΔESP increased at 40 cm by 9 %, 7 % at 60 cm and 1 % at 80 cm. Compared to December 2002, soil ESP for February 2003 increased at all depths. At 10 cm depth, ΔESP had risen approximately 10 %, 7 % at 20 cm, 5 % at 40 cm, 1 % at 60 cm and 5 % at 80 cm.

Compared to February 2003, soil ESP for April 2003 decreased at all depths. At 10 cm depth, ΔESP had fallen approximately 11 %, 9 % at 20 cm, 18 % at 40 cm, 10 % at 60 cm and 5 % at 80 cm. April 2003 to June 2003 exhibited an increase in soil ESP at all depths. At a depth of 10 cm, ΔESP had increased by approximately 7 %, 4 % at 20 cm, 12 % at 40 cm and 5 % at 60 cm depth. Negligible increase was observed at 80 cm depth for this period. Soil ESP during August 2003 decreased at ≤ 60 cm depths and increased at a depth of 80 cm. At 10 cm depth, ΔESP decreased by approximately 1%, negligible decrease at 20 cm, 7 % at 40 cm and 4% at 60 cm. Soil ΔESP increased at a depth of 80 cm by 1 %.

During October 2003, the 10 cm, 20 cm and 80 cm depths recorded decreases, while the 40 cm and 60 cm depths recorded increases in soil ESP. At 10 cm depth, ΔESP decreased by approximately 5 %, with a negligible decrease at 20 cm and 1 % decrease at 80 cm. Soil ΔESP increased at 40 cm and 60 cm by approximately 8 %.

The change in sodium (cmol(+)/kg) shows both accumulation and leaching over time and indicates that downward sodium movement is being maintained. The values of ESP and ΔESP are much larger in magnitude than the change in sodium (cmol(+)/kg) and the significance is that relatively large variations in ESP do not necessarily reflect changes in soluble sodium within the soil profile. This is particularly evident for February 2003 and June 2003.
Site 5 – S5

Figure 5.25 shows soil ESP, change in ESP (ΔESP) and change in sodium (cmol(+)/kg) for S5 between February 2002 and October 2003. ESP values at 10 cm ranged from approximately 10 % to 15 % for the entire monitoring period. At 20 cm, 40 cm, 60 cm and 80 cm, ESP values ranged from 12 % to 20 %, 15 % to 27 %, 16 % to 34 % and 22 % to 43 % respectively. ESP values for February were designated zero to determine subsequent increases/decreases in ΔESP for April 2002 results. The major observation is that there was significant sodium movement (as cmol(+)/kg) throughout the soil profile. Both June 2002 and August 2003 recorded increases in sodium (cmol(+)/kg, which are also reflected in the ΔESP.

The columns in April 2002 represent the ΔESP for February 2002 to April 2002. For this period, soil ESP increased at 80 cm, although decreased at 10 cm, 20 cm, 40 cm and 60 cm. At 10 cm depth, ΔESP had increased approximately 1 % and increased 3 % at 80 cm. At 20 cm depth, ΔESP had decreased by 2 % at 20 cm, 4 % at 40 cm and 3 % at 60 cm. This indicates a net sodium flux in favour of accumulation at a depth of 80 cm.

Compared to April 2002 ΔESP values all depths recorded an increase in ESP in June 2002. At 10 cm depth, ΔESP had increased by approximately 2.5 %, 7 % at 20 cm, 5 % at 40 cm, 7.5 % at 60 cm and 8 % at 80 cm. Except at 10 cm, all depths recorded decreases in soil ESP for August 2002. Negligible increase in ESP was observed at 10 cm depth for this period. ΔESP decreased by approximately 3 % at 20 cm, 4 % at 40 cm, 3.5 % at 60 cm and 7.5 % at 80 cm. Compared to August 2002, soil ESP for October 2002 decreased at a depth of 10 cm and 20 cm and remained unchanged at 40 cm, 60 cm and 80 cm. At 10 cm depth, ΔESP had fallen approximately 3 % and 5 % at 20 cm depth.

Soil ESP for the October 2002 to December 2002 remained relatively unchanged a depth of 10 cm and 60 cm. Soil ESP decreased at 80 cm by approximately 6 %. At 20 cm depth, ΔESP increased by approximately 4 % and 6 % at 60 cm. Soil ΔESP decreased at 80 cm by 6 %. Relative to December 2002, soil ESP for February 2003 had increased at a depth of 10 cm and 20 cm by approximately 4 % and 2.5 % respectively,
Figure 5.25: Soil ESP, ΔESP and change in sodium (cmol(+) / kg) at S5 (February 2002 – October 2003)
but decreased at 40 cm, 60 cm and 80 cm. At 40 cm depth, $\Delta ESP$ had decreased by approximately 6.5 %, 6 % at 60 cm and 3 % at 80 cm.

Relative to February 2003, soil ESP for April 2003 decreased at all depths. At 10 cm depth, $\Delta ESP$ had fallen approximately by 5 %, 5 % at 20 cm, 2.5 % at 40 cm, 5 % at 60 cm and 2 % at 80 cm. April 2003 to June 2003 exhibited an increase in soil ESP at a depth of 10 cm, 20 cm and 80 cm and a decrease in soil ESP at 40 cm and 60 cm depth. At 10 cm and 20 cm depth, $\Delta ESP$ had increased by approximately 1 % and decreased by 3 % at 40 cm and 5 % at 60 cm. $\Delta ESP$ increased by approximately 7 % at a depth of 80 cm, indicating sodium accumulation.

Soil ESP during August 2003 increased at all depths. At 10 cm depth, $\Delta ESP$ increased by approximately 2.5 %, 3 % at 20 cm, 12 % at 40 cm, 22 % at 60 cm and 11 % at 80 cm. During October 2003, the 20 cm, 40 cm, 60 cm and 80 cm depths recorded decreases, while the 10 cm depth recorded an increase in soil ESP. At 10 cm depth, $\Delta ESP$ increased by approximately 2 %. Soil $\Delta ESP$ decreased at 20 cm by a negligible amount, by approximately 8 % at 60 cm, and approximately 11 % at depths of 60 cm and 80 cm.

All sites show that ESP increased with increased depth and this was a common result for all irrigated and non-irrigated sites. Changes in sodium (cmol(+)/kg) at S5 were much larger than at S1. This was due to the presence of sandy clay at 30 cm and higher $\theta$ values at S5 which promoted advective solute movement through the soil profile, at least to the sampled depth of 80 cm.

**Site 8 – S8**

Figure 5.26 shows soil ESP, change in ESP ($\Delta ESP$) and change in sodium (cmol(+)/kg) for S8 between February 2002 and October 2003. ESP values at a depth of 10 cm ranged between 4 % and 13 % for the entire monitoring period. At 20 cm, 40 cm, 60 cm and 80 cm, ESP values ranged from 5 % to 15 %, 5 % to 17 %, 7 % to 33 % and 13 % to 35 % respectively. Compared to February 2002, $\Delta ESP$ decreased at all recorded
Figure 5.26: Soil ESP, $\Delta$ESP and change in sodium (cmol(+) / kg) at S8 (February 2002 – October 2003)
depths during the April 2002 sampling period. At 10 cm depth, $\Delta$ESP had decreased by 4 % at 10 cm, 8 % at 20 cm, 10 % at 40 cm, 15 % at 60 cm and 20 % at 80 cm. The loss of sodium over all depths indicates a leaching trend for this soil sampling period.

Compared to April 2002 $\Delta$ESP values increased at all recorded depths in June 2002. $\Delta$ESP had increased by approximately 1 % at 10 cm and 20 cm, 6 % at 40 cm, 10 % at 60 cm and 12 % at 80 cm. Compared to June 2002 $\Delta$ESP values at all depths recorded further increases in ESP in August 2002, indicating net sodium accumulation. At 10 cm depth, $\Delta$ESP had increased by approximately 3 %, 7 % at 20 cm, 4 % at 40 cm, 1 % at 60 cm and 3 % at 80 cm, indicating a net sodium accumulation from the previous sampling period.

Relative to August 2002, negligible change in $\Delta$ESP was observed at a depth of 40 cm in October 2002. However, an approximate 2 % decrease occurred at a depth of 10 cm and 20 cm, with relatively large increases having occurred at depths of 60 cm (12 %) and 80 cm (7 %). Relative to October 2002, negligible change in $\Delta$ESP at depths of 10 cm and 40 cm were recorded in December 2002, while soil at 60 cm and 80 cm depths decreased. $\Delta$ESP decreased by approximately 8 % at both 60 cm and 80 cm depths during this period, indicating that most of the sodium accumulated during the previous sampling period had been leached to lower depths.

Soil ESP for February 2003 relative to December 2002 increased at all depths except at 10 cm. $\Delta$ESP fell by 8 % at 10 cm while at 20 cm depth, $\Delta$ESP increased by approximately 2 %, 17 % at 40 cm, 9 % at 60 cm and 8 % at 80 cm depth. For April 2003, soil ESP decreased at all depths. At 10 cm depth, $\Delta$ESP had decreased by a negligible amount. At 20 cm, $\Delta$ESP decreased by approximately 8 %, 15 % at 40 cm and 1 % at both 60 cm and 80 cm depth.

Relative to April 2003, soil ESP for June 2003 increased at 10 cm and 20 cm depth and decreased at 40 cm, 60 cm and 80 cm depth. At 10 cm depth, $\Delta$ESP increased approximately 6 % and 3 % at 20 cm. At a depth of 40 cm, $\Delta$ESP decreased by less than 1 %, 13 % at 60 cm and 5 % at 80 cm. June 2003 to August 2003 exhibited an increase in soil ESP at 60 cm and a decrease at 10 cm, 20 cm, 40 cm and 80 cm depth. At 10 cm depth, $\Delta$ESP had decreased by approximately 5 %, 6 % at 20 cm, 2 % at 40 cm and 2.5
% at 80 cm. ΔESP had increased at 60 cm by approximately 8 %. Soil ESP during October 2003 increased at all depths except 60 cm, which underwent a 1 % decrease. At 10 cm depth, ΔESP increased by approximately 3 %, 3.5 % at 20 cm, 11 % at 40 cm and 3 % at 80 cm, indicating further sodium accumulation.

The change in sodium (cmol(+)/kg) shows both accumulation and leaching over time while temporal ΔESP values indicated that downward sodium movement occurred. The values of ESP and ΔESP are much larger in magnitude than the change in sodium (cmol(+)/kg) and as with S5, the significance is that relatively large variations in ESP do not necessarily reflect changes in soluble sodium within the soil profile. This is particularly evident for June 2002, February 2003 and June 2003. S8 remained near-saturation from February 2002 to October 2003, while S1, a profile of similar texture, had θ values near wilting point at depths of greater than 20 cm for most of the same period. However, both show minimal change in sodium (cmol(+)/kg) between sample periods at all depths, despite the difference in θ between S8 and S1 (refer Figures 5.14 and 5.9 respectively).

Duplicate sampling was performed at different depths at S8 during October 2002 and results are shown in Table 5.5.

**Table 5.5: Duplicate sampling during October 2002 at S8 to determine at-a-site variability.**

<table>
<thead>
<tr>
<th>Date</th>
<th>Site</th>
<th>Depth (cm)</th>
<th>θ</th>
<th>pHw</th>
<th>EC</th>
<th>1:5 (dS/m)</th>
<th>Exch. Cat. 1M NH4Cl</th>
<th>cmol(+)/kg</th>
<th>%</th>
</tr>
</thead>
<tbody>
<tr>
<td>Oct-02</td>
<td>8</td>
<td>10</td>
<td>11.8</td>
<td>6.04</td>
<td>0.253</td>
<td>8.474</td>
<td>1.159</td>
<td>0.834</td>
<td>0.268</td>
</tr>
<tr>
<td>Oct-02</td>
<td>8B</td>
<td>10</td>
<td>12.5</td>
<td>6.03</td>
<td>0.273</td>
<td>7.875</td>
<td>1.242</td>
<td>0.796</td>
<td>0.243</td>
</tr>
<tr>
<td>Oct-02</td>
<td>8</td>
<td>20</td>
<td>15.8</td>
<td>6.22</td>
<td>0.232</td>
<td>6.172</td>
<td>0.761</td>
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<td>0.292</td>
</tr>
<tr>
<td>Oct-02</td>
<td>8B</td>
<td>20</td>
<td>15.0</td>
<td>6.18</td>
<td>0.223</td>
<td>4.540</td>
<td>1.031</td>
<td>0.767</td>
<td>0.345</td>
</tr>
<tr>
<td>Oct-02</td>
<td>8</td>
<td>40</td>
<td>13.5</td>
<td>6.24</td>
<td>0.318</td>
<td>3.953</td>
<td>0.601</td>
<td>0.835</td>
<td>0.261</td>
</tr>
<tr>
<td>Oct-02</td>
<td>8B</td>
<td>40</td>
<td>10.3</td>
<td>6.27</td>
<td>0.196</td>
<td>2.867</td>
<td>0.312</td>
<td>0.681</td>
<td>0.175</td>
</tr>
<tr>
<td>Oct-02</td>
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<td>60</td>
<td>11.4</td>
<td>6.15</td>
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<td>1.017</td>
<td>0.125</td>
</tr>
<tr>
<td>Oct-02</td>
<td>8B</td>
<td>60</td>
<td>10.2</td>
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<td>0.667</td>
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<tr>
<td>Oct-02</td>
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<td>12.5</td>
<td>6.21</td>
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<td>1.714</td>
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<td>1.107</td>
<td>0.106</td>
</tr>
<tr>
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<td>80</td>
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<td>6.39</td>
<td>0.158</td>
<td>1.102</td>
<td>0.130</td>
<td>0.636</td>
<td>0.069</td>
</tr>
</tbody>
</table>
The variation in ESP between duplicate soil samples is shown to be much less than the temporal variations shown in Figure 5.26. This shows that the spatial variation seen in soil ESP is much less than at-a-site variation observed over time. However, the relative composition with respect to Ca\(^{2+}\), Mg\(^{2+}\) and Na\(^{+}\) shows much greater variation. This will be investigated later in this thesis. Also, as with other sites, soil ESP at S8 also increased with depth on each sampling occasion.

**Site 9 – S9 (non-irrigated site)**

Site 9 received rainfall only and comprised of both loamy sand and sandy clay (see Figure 5.8). Therefore, changes in soil ESP were a consequence of rainfall frequency and not irrigation frequency. Figure 5.27 shows soil ESP, change in ESP (ΔESP) and change in sodium (cmol(+)/kg) for S9 between February 2002 and October 2003. Comparisons between irrigated and non-irrigated sites are made in Section 5.3.5.

At S9, ESP values at 10 cm ranged between 5 % and 22 % for the entire monitoring period. At 20 cm, 40 cm, 60 cm and 80 cm, ESP values ranged from 4 % to 34 %, 4 % to 42 %, 17 % to 30 % and 28 % to 47 % respectively. Even though S9 is not irrigated, soil ESP still increased with depth. However, the change in sodium (cmol(+)/kg) between depths and over time was shown to be highly variable, particularly in the sandy clay horizons at depths greater than 40 cm.

At 10 cm depth, ΔESP increased by approximately 4 % from February 2002 to April 2002. At 20 cm depth, ΔESP had decreased by 6 % and by negligible amounts at 40 cm, 60 cm and 80 cm depths. Compared to April 2002 ΔESP values all depths recorded a decrease in ESP in June 2002. At 10 cm depth, ΔESP had decreased by approximately 4 %. ΔESP decreased by negligible amounts at 40 cm, 60 cm and 80 cm depths.

Compared to June 2002 ΔESP values, August 2002 ΔESP values increased at 40 cm and 60 cm and decreased at 10 cm, 20 cm and 80 cm. At 10 cm depth, ΔESP had decreased by approximately 5 %, 3 % at 20 cm and 4 % at 80 cm. At 40 cm, ΔESP had increased by 11 % and less than 1 % at 60 cm depth. All depths recorded decreases in soil ESP for October 2002, although at 80 cm this decrease is negligible. ΔESP had
decreased by approximately 3% at 10 cm, 10% at 20 cm, 22% at 40 cm and 11% at 60 cm.

Figure 5.27: Soil ESP, ΔESP and change in sodium (cmol(+)/kg) at S9 (February 2002 – October 2003)
Relative to October 2002, depths at 10 cm and 20 cm recorded decreases in soil ESP for December 2002, while values at 40 cm, 60 cm and 80 cm depths increased. ΔESP had decreased by approximately 5% at 10 cm and 2% at 20 cm. ΔESP increased by approximately 2% at 40 cm, 6% at 60 cm and 5% at 80 cm depths during this period.

Soil ESP for February 2003 relative to December 2002 decreased at all depths except at 10 cm. ΔESP increased by approximately 8% at 10 cm while at 20 cm depth, ΔESP decreased by approximately 8%, 5% at 40 cm, 8% at 60 cm and 5% at 80 cm depth. For April 2003, soil ESP increased at all depths except at 40 cm. At 10 cm depth, ΔESP had increased by 9%, 6% at 20 cm, 1% at 60 cm and 8% at 80 cm. ΔESP decreased by approximately 2% at 40 cm.

Relative to April 2003, soil ESP for June 2003 decreased at 10 cm, 20 cm and 40 cm but increased at 60 cm and 80 cm. At 10 cm depth, ΔESP decreased approximately 3%, 8% at 20 cm and 8.5% at 40 cm. ΔESP increased by 8% at 60 cm and 12% at 80 cm. June 2003 to August 2003 exhibited an increase in soil ESP at all depths except at 80 cm, which underwent a decrease of 18%. At 10 cm depth, ΔESP had increased by approximately 21%, 8% at 20 cm, 24% at 40 cm and 3% at 60 cm. Soil ESP during October 2003 decreased at all depths except 80 cm, which underwent a negligible increase. At 10 cm depth, ΔESP decreased by approximately 30%, 4% at 20 cm, 14% at 40 cm and 5% at 60 cm. In fact, all sites with sandy clay present at depth exhibited greater sodium accumulation/leaching than sites with just loamy sands.

**Site 10 – S10**

Figure 5.28 shows soil ESP, change in ESP (ΔESP) and change in sodium (cmol(+)/kg) for S10 between February 2002 and October 2003. ESP values at 10 cm ranged between 14% and 27% for the entire monitoring period. At 20 cm, 40 cm, 60 cm and 80 cm depths, ESP values ranged from 15% to 33%, 23% to 52%, 24% to 42% and 25% to 35% respectively. From February 2002 to April 2002, soil ESP increased by a negligible amount at 10 cm and 40 cm depths, with a negligible decrease
observed at 80 cm depth. At 20 cm and 40 cm depths, \( \Delta \text{ESP} \) increased by approximately 4%.

Compared to April 2002 \( \Delta \text{ESP} \) values all depths recorded further increases in ESP for June 2002. At 10 cm depth, \( \Delta \text{ESP} \) had increased by approximately 3.5%, 10% at 20 cm, 14% at 40 cm, 1% at 60 cm and less than 1% at 80 cm. Compared to June 2002 \( \Delta \text{ESP} \) values, August 2002 \( \Delta \text{ESP} \) values decreased at 40 cm and increased at 10 cm, 20 cm, 60 cm and 80 cm. At 10 cm depth, \( \Delta \text{ESP} \) increased by approximately 4.5%, 6% at 20 cm, 11% at 60 cm and 2% at 80 cm. At 40 cm, \( \Delta \text{ESP} \) decreased by 3%.

Soil ESP for October 2002 decreased at all depths except at 80 cm. At 10 cm, \( \Delta \text{ESP} \) decreased by 4%, 6% at 20 cm, 18% at 40 cm and 13% at 60 cm. At 80 cm, \( \Delta \text{ESP} \) increased by 3%. Relative to October 2002, depths at 20 cm and 60 cm recorded increases in soil ESP for December 2002, while \( \Delta \text{ESP} \) at 40 cm and 80 cm decreased. No change in soil ESP was observed at 10 cm. \( \Delta \text{ESP} \) had increased by approximately 5% at 20 cm and 1% at 60 cm. \( \Delta \text{ESP} \) decreased by less than 1% at 40 cm and 5% at 80 cm depths during this period. Soil ESP for December 2002 relative to February 2003 increased at all depths. \( \Delta \text{ESP} \) increased by approximately 9% at 10 cm, 9% at 20 cm, 24% at 40 cm, 5% at 60 cm and 4% at 80 cm.

Relative to soil ESP in February 2003, soil ESP for April 2003 decreased at all depths. At 10 cm depth, \( \Delta \text{ESP} \) had decreased by 4%, 13% at 20 cm, 18% at 40 cm, 2% at 60 cm and 4% at 80 cm. Relative to April 2003, soil ESP for June 2003 decreased at 10 cm, 40 cm, 60 cm and 80 cm, although increases at 20 cm. At 10 cm depth, \( \Delta \text{ESP} \) decreased approximately 5%, 9% at 40 cm, 10% at 60 cm and 3.5% at 80 cm. \( \Delta \text{ESP} \) increased by 8% at 20 cm.

June 2003 to August 2003 exhibited an increase in soil ESP at depths of 40 cm and 60 cm, and a decrease at depths of 10 cm, 20 cm and 80 cm. At 40 cm, \( \Delta \text{ESP} \) increased to approximately 3% and 4% at 60 cm. At 10 cm depth, \( \Delta \text{ESP} \) decreased by approximately 4%, 12% at 20 cm and 2% at 80 cm. Soil ESP during October 2003 increased at all depths except 40 cm, which underwent a decrease of approximately 4%.
Figure 5.28: Soil ESP, ΔESP and change in sodium (cmol(+)/kg) at S10 (February 2002 – October 2003)
At 10 cm depth, $\Delta$ESP increased by approximately 5%, 7% at 20 cm, 2% at 60 cm and 5% at 60 cm.

Change in sodium (cmol(+)/kg) for each sampling occasion showed significant variation. S10 had sandy clay at depths greater than 20 cm (refer Figure 5.8) and remained near-saturated for most of the study period (see Figure 5.15). Due to the predominance of sandy clay at S10, a greater cation exchange capacity (CEC) would be expected, particularly with excess sodium in the permeating waters.

On several occasions during 2003, the change in sodium (cmol(+)/kg) did not reflect $\Delta$ESP. This issue seems to arise when sodium accumulation and leaching occur at different depths within the same profile at a given time. For example, April 2003, June 2003 and August 2003 change in sodium (cmol(+)/kg) results at all depths are almost opposite to $\Delta$ESP results for the same occasions. The fact that $\Delta$ESP values between sampling occasions do not reflect actual sodium accumulation/leaching is of concern, considering soil ESP is an established soil property in the literature for determining variations in soil sodium over time, and will be further discussed in chapter 6.

**Site 14 – S14 (non-irrigated site)**

Figure 5.29 shows soil ESP, change in ESP ($\Delta$ESP) and change in sodium (cmol(+)/kg) for S14 between February 2002 and October 2003. S14 was a non-irrigated site that received rainfall only and comprised of loamy sand to the sampled depth of 80 cm. Therefore, any changes in soil ESP ($\Delta$ESP) are more likely to reflect rainfall frequency, rather than irrigation frequency. Comparisons between irrigated and non-irrigated sites are made in Section 5.3.5.

ESP values at 10 cm ranged from approximately 2% to 14% for the entire monitoring period. At 20 cm, 40 cm, 60 cm and 80 cm, ESP values ranged from 3% to 20%, 5% to 28%, 3% to 29% and 4% to 30% respectively. The columns in April 2002 represent the $\Delta$ESP for February 2002 to April 2002. All depths recorded a less than 1% decrease in ESP during this period.
Figure 5.29: Soil ESP, ΔESP and change in sodium (cmol(+)/kg) at S14 (February 2002 – October 2003)
Compared to April 2002 ΔESP values, all depths increased in ESP in June 2002. At 10 cm depth, ΔESP had risen by approximately 11 %, 15 % at 20 cm, 23 % at 40 cm, 20 % at 60 cm and 23.5 % at 80 cm.

Soil ESP values for August 2002 are similar to June 2002 values, as ΔESP does not exceed 4 %. Depths at 20 cm, 40 cm, 60 cm and 80 cm recorded increased ESP (ΔESP = < 4 %), while at 10 cm, a 2 % decrease was observed. Compared to August 2002, soil ESP for October 2002 decreased significantly. At 10 cm depth, ΔESP had fallen approximately 5 %, 10 % at 20 cm, 18 % at 40 cm, 17 % at 60 cm and 19 % at 80 cm. Soil ESP during December 2002 increased at 10 cm, 40 cm, 60 cm and 80 cm depths, all at ΔESP values less than 4 %. At 20 cm, soil ESP decreased to 3 %. Compared to December 2002, soil ESP for February 2003 increased at all depths. At 10 cm depth, ΔESP had risen less than 1 %, approximately 5 % at 20 cm, 3 % at 40 cm and 1 % at 60 cm and 80 cm.

Compared to February 2003, soil ESP for April 2003 decreased at 10 cm, 20 cm and 80 cm depth, whereas had fallen at 40 cm and 60 cm. ΔESP increased at 10 cm by 5 %, 10 % at 20 cm and 4 % at 80 cm. ΔESP decreased at 40 cm by approximately 9 % and 16 % at 60 cm. Relative to April 2003 June 2003 exhibited an decrease in soil ESP at 40 cm and 60 cm and increased soil ESP at 10 cm, 20 cm and 80 cm. At 10 cm depth, ΔESP increased by approximately 8 %, 13 % at 20 cm and 10 % at 80 cm. ΔESP decreased at 40 cm by approximately 9 % and 15 % at 60 cm.

Soil ESP during August 2003 decreased at all depths except at 40 cm, which increased approximately 5 %. At 10 cm depth, ΔESP decreased by approximately 7 %, 12 % at 20 cm, 10 % at 60 cm and 3 % at 80 cm. During October 2003, the 10 cm, 20 cm, 60 cm and 80 cm depths recorded increases in soil ESP, while the 40 cm depth recorded a decrease in soil ESP (ΔESP = 4 %). At 10 cm depth, ΔESP increased by less than 1 %, 10 % at 20 cm, 15 % at 60 cm and 2 % at 80 cm.

The change in sodium (cmol(+)/kg) reflects ΔESP values, particularly in June 2002 and October 2002, although the change in sodium (cmol(+)/kg values are smaller in magnitude. This is in contrast to S9, where ΔESP did not necessarily relate to change
in sodium (cmol(+)/kg). From April 2003 onwards, sodium accumulation and leaching are dominant in the uppermost 40 cm.

5.3.4 Soil analyses – Average temporal soil ESP and ΔESP and comparison of temporal soil ESP between similar sites

Figure 5.30 shows the average ESP and average change in ESP (ΔESP) for all sites in the woodlot (S1, S5, S8 and S10) between February 2002 and October 2003, while Table 5.6 gives the 95 % confidence interval for the presented values. Average ESP values at 10 cm ranged from approximately 11 % to 16 % for the entire monitoring period. At 20 cm, 40 cm, 60 cm and 80 cm, average ESP values ranged from 13 % to 24 %, 11 % to 29 %, 18 % to 32 % and 27 % to 35 % respectively.

For April 2002, zero change in average ESP was recorded at 10 cm depth. The 20 cm depth recorded an increase in average ESP of approximately 1 %. The 40 cm, 60 and 80 cm depths recorded a decrease in average ESP by 1 %, 4 % and 1.5 % respectively. Compared to April 2002 average ΔESP values, all depths increased in average ESP in June 2002. At 10 cm depth, average ΔESP had risen by approximately 2.5 %, 5 % at 20 cm, 9 % at 40 cm, 8.5 % at 60 cm and 5 % at 80 cm. Average soil ESP values for August 2002 are similar to June 2002 values, although average ESP does increase slightly. Depths at 10 cm, 20 cm, 40 cm, 60 cm and 80 cm recorded increased average ESP (average ΔESP = < 4 %), indicating a slow leaching response from the previous soil sampling period.

Compared to August 2002, average soil ESP for October 2002 decreased significantly compared to other soil sampling periods, particularly at depths of 20 cm and 40 cm. In addition, this was the first time that soil sodium decreased at all recorded depths, as the previous two soil sampling periods indicated sodium accumulation. At 10 cm depth, average ΔESP had fallen approximately 4 %, 7.5 % at 20 cm, 8 % at 40 cm, 3 % at 60 cm and 3 % at 80 cm.
Figure 5.30: Average ESP and average ΔESP for all sites within the woodlot (S1, S5, S8, S10) (February 2002 – October 2003)
Table 5.6: 95% Confidence Intervals for soil ESP values – top graph of Figure 5.30.

<table>
<thead>
<tr>
<th></th>
<th>10 cm</th>
<th>20 cm</th>
<th>40 cm</th>
<th>60 cm</th>
<th>80 cm</th>
</tr>
</thead>
<tbody>
<tr>
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<td>5.28</td>
<td>6.47</td>
</tr>
<tr>
<td>Apr-02</td>
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<td>6.59</td>
<td>8.85</td>
<td>9.22</td>
<td>9.36</td>
</tr>
<tr>
<td>Jun-02</td>
<td>5.45</td>
<td>9.57</td>
<td>12.11</td>
<td>6.74</td>
<td>7.30</td>
</tr>
<tr>
<td>Aug-02</td>
<td>7.05</td>
<td>10.00</td>
<td>14.14</td>
<td>9.75</td>
<td>9.16</td>
</tr>
<tr>
<td>Oct-02</td>
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<td>4.55</td>
<td>5.58</td>
<td>4.30</td>
<td>3.82</td>
</tr>
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<td>6.73</td>
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<td>1.51</td>
</tr>
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<td>13.32</td>
<td>5.01</td>
<td>5.40</td>
</tr>
<tr>
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<td>5.70</td>
<td>3.44</td>
<td>2.26</td>
<td>1.58</td>
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</tbody>
</table>

Average soil ESP during December 2002 increased at 10 cm, 20 cm, 40 cm and 60 cm by less than 4%. At 80 cm, average soil ESP decreased by 4%. Compared to December 2002, average soil ESP for February 2003 increased at all depths. At 10 cm depth, average ΔESP had risen approximately 4%, approximately 5% at 20 cm, 10% at 40 cm and 2.5% at 60 cm and 80 cm. Compared to February 2003, average soil ESP for April 2003 decreased at all depths and shows the largest decrease in soil sodium for the entire study period at this site. Average ΔESP decreased at 10 cm by approximately 6%, 8% at 20 cm, 13% at 40 cm, 5% at 60 cm and 4% at 80 cm.

Relative to April 2003, average soil ESP in June 2003 exhibited a decrease at 60 cm and 80 cm and increased average soil ESP at 10 cm and 20 cm. At 10 cm depth, average ΔESP increased by approximately 1.5% and 4% at 20 cm. Average ΔESP decreased at 60 cm by approximately 6.5% and 1% at 80 cm. The largest change in ΔESP was recorded at 40 cm, which increased by approximately 12.5%. Average soil ESP during August 2003 decreased at depths of 10 cm, 20 cm and 40 cm by approximately 2% and 4% and 7% respectively. Average ΔESP increased by approximately 7% at 60 cm and 2.5% at 80 cm. During October 2003, the 10 cm, 20
cm and 40 cm depths recorded increases in average soil ESP, while the 60 cm and 80 cm depths recorded a decrease in average soil ESP (average $\Delta$ESP = $< 1 \%$). At 10 cm depth, average $\Delta$ESP increased by 1.5 %, 2.5 % at 20 cm and 2.5 % at 40 cm.

The alternating accumulation/leaching (short-term) is particularly evident from June 2003 to October 2003 and is indicated by the gain/loss between sampling periods and at recorded depths. Accumulation/leaching (long-term) were particularly evident during 2002, where the soil sodium gain/loss is consistent for all depths on each sampling occasion. That is, soil sodium decreases at all recorded depths then increases at all recorded depths the following soil sampling period.

Table 5.7 shows the average soil ESP as a function of depth between irrigated and non-irrigated sites and in context of similar soil profiles (S1+S8 v S9 and S5+S10 v S9). With respect to sample depth and similar soil profiles, the averages presented indicate that soil ESP is generally greater at sites receiving effluent irrigation.

Table 5.7: Average soil ESP for Irrigated/Non-Irrigated Loamy Sand and Irrigated/Non-Irrigated Loamy Sand (A horizon)/Sandy Clay (B horizon)

<table>
<thead>
<tr>
<th>Irrigated Loamy Sand</th>
<th>10 cm</th>
<th>20 cm</th>
<th>40 cm</th>
<th>60 cm</th>
<th>80 cm</th>
</tr>
</thead>
<tbody>
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<td>22.4</td>
<td>23.4</td>
<td>23.1</td>
<td>23.4</td>
<td>27.5</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Non-Irrigated Loamy Sand</th>
<th>10 cm</th>
<th>20 cm</th>
<th>40 cm</th>
<th>60 cm</th>
<th>80 cm</th>
</tr>
</thead>
<tbody>
<tr>
<td>S14</td>
<td>13.1</td>
<td>13.8</td>
<td>17.1</td>
<td>18.1</td>
<td>18.2</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Irrigated Sandy Clay</th>
<th>10 cm</th>
<th>20 cm</th>
<th>40 cm</th>
<th>60 cm</th>
<th>80 cm</th>
</tr>
</thead>
<tbody>
<tr>
<td>S5+S10</td>
<td>15.0</td>
<td>16.3</td>
<td>22.4</td>
<td>23.5</td>
<td>23.6</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Non-Irrigated Sandy Clay</th>
<th>10 cm</th>
<th>20 cm</th>
<th>40 cm</th>
<th>60 cm</th>
<th>80 cm</th>
</tr>
</thead>
<tbody>
<tr>
<td>S8</td>
<td>16.8</td>
<td>18.2</td>
<td>22.7</td>
<td>28.2</td>
<td>32.7</td>
</tr>
</tbody>
</table>

Table 5.7 highlights that soil ESP generally increases with depth and that $\text{Na}^+$ - rich waters increase $\text{Na}^+$ in the soil profile over time when compared to similar sites not receiving $\text{Na}^+$ - rich waters.

5.3.5 Temporal variation in cations (Ca$^{2+}$, Mg$^{2+}$, $\text{Na}^+$ and $\text{K}^+$), cation exchange capacity (CEC) and the influence of clay mineralogy

Figures 5.30 to 5.35 show changes in cations (Ca$^{2+}$, Mg$^{2+}$, $\text{Na}^+$ and $\text{K}^+$ in cmol(+)/kg) and cation exchange capacity (CEC) with depth for S1, S5, S8, S9, S10 and
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S14 respectively between February 2002 and October 2003. The main observation for all sites is the varying degree of cation flux over time and depth. Sites with sandy clay horizons (S5, S10 and the non-irrigated site S9) all have greater cation exchange properties, particularly with respect to magnesium (Mg$^{2+}$).

Figure 5.31 shows the relative change in cations and CEC for S1 at depths of 10 cm, 20 cm, 40 cm, 60 cm and 80 cm between February 2002 and October 2003. The columns in April 2002 represent the change in cations and CEC for February 2002 to April 2002. The change in individual cations and CEC reflect the molar concentration lost or gained from one monitoring period to another. At S1, the largest cation flux is most prevalent in the uppermost 20 cm of the soil profile. At depths of less than 40 cm, negligible cation flux is observed and this corresponds to the leached A2 horizons for S1 shown in Figure 5.8.

In Figure 5.32 the dominant cation exchanged within the soil profile at S5 was Mg$^{2+}$. The accumulation/leaching of excess Mg$^{2+}$ can be observed at all depths, however the source of Mg$^{2+}$ cannot be attributed to effluent irrigation. This is indicated by the presence of excess Mg$^{2+}$ at all depths at the non-irrigated site S9 (Figure 5.34).

In Figure 5.33 shows the relative difference in cations and CEC for S8. S8 was dosed with lime in 1994 and the presence of accumulated/leached Ca$^{2+}$ at all depths can be seen. Conspicuous in its absence, Na$^{+}$ flux appears to be minimal. However, ΔESP values from Figure 5.25 suggest the opposite, as ΔESP values at S8 were some of the largest variations of any site. It can be seen from Figure 5.33 that the variation in both Ca$^{2+}$ and Mg$^{2+}$ are the cause of the large ΔESP variations in Figure 5.25.

Figure 5.34 shows the relative difference in cations and CEC for S9. S9 was a non-irrigated site that had sandy clay at a depth of 40 cm. Temporal cation exchange was most prevalent at depths of 60 cm and greater, with Mg$^{2+}$ again being the dominant cation involved. At depths of 40 cm and less, the largest variation occurred from June – August 2003.
Figure 5.31: Change in molar concentration of cations (Ca$^{2+}$, Mg$^{2+}$, Na$^+$ and K$^+$) in cmol(+)/kg and cation exchange capacity (CEC) with depth for S1
Figure 5.32: Change in molar concentration of cations (Ca$^{2+}$, Mg$^{2+}$, Na$^+$ and K$^+$) in cmol(+)/kg and cation exchange capacity (CEC) with depth for S5.
Figure 5.33: Change in molar concentration of cations (Ca$^{2+}$, Mg$^{2+}$, Na$^{+}$ and K$^{+}$ in cmol(+)/kg) and cation exchange capacity (CEC) with depth for S8.
Figure 5.34: Change in molar concentration of cations (Ca$^{2+}$, Mg$^{2+}$, Na$^{+}$ and K$^{+}$) and cation exchange capacity (CEC) with depth for S9.
Figure 5.35: Change in molar concentration of cations (Ca$^{2+}$, Mg$^{2+}$, Na$^{+}$ and K$^{+}$) in cmol(+)/kg and cation exchange capacity (CEC) with depth for S10
Figure 5.36: Change in molar concentration of cations (Ca$^{2+}$, Mg$^{2+}$, Na$^+$ and K$^+$) in cmol(+)/kg and cation exchange capacity (CEC) with depth for S14.
Figure 5.35 shows the relative difference in cations and CEC for S10. S10 was similar to S5 in that significant cation exchange occurred at all depths, although S10 contains higher Mg$^{2+}$ concentrations. When compared to the non-irrigated site S9, the irrigated site S10 shows excess cations are present, indicating either nutrient deficient soils adjacent to the woodlot or S10 has accumulated nutrient through irrigation over time. This is evident from the greater cation flux over time in the uppermost 20 cm of the soil profile at S10, than at S9.

Figure 5.36 shows the relative difference in cations and CEC for S14. Small variations in cation exchange can be observed in the uppermost 20 cm, with insignificant cation flux at lower depths. It must be noted that S14 consisted of leached loamy sands and the CEC of the soil was very low.

Cation flux is shown to be highly dynamic at all depths at irrigated sites with sandy clay B-horizons (S5 and S10). In contrast, irrigated sites with only loamy sand to the sampled depth of 80 cm (S1 and S8), showed that cation flux was dominant in the upper 40 cm of the soil profile, most likely as a result of excess cations supplied by irrigated effluent. Cation exchange at the non-irrigated site S9 (with sandy clay B-horizon) showed the majority of cation flux occurred at less than 40 cm. Cation flux at the non-irrigated site S14 (loamy sand to 80 cm) showed the majority of variations occurred nearer the surface.

Several relationships were observed when soil ESP was plotted against cation ratios for all soil samples. Figure 5.37 shows the ratio of soil ESP versus molar Ca/Mg, Na/Mg and Ca/Na. Soil ESP versus Ca/Mg ratio for loamy sands is generally higher and shows more variability than sandy clay samples. The Ca/Mg ratio trends towards a decrease as soil ESP increases which was most likely due to the influence of excess Na$^+$ at high soil ESP values.

Soil ESP versus Na/Mg ratio for loamy sands again shows more variability than sandy clay samples however values for loamy sands were generally lower than other cation ratios. The Na/Mg ratio trends towards an increase as soil ESP increases, suggesting that Mg$^{2+}$ plays a significant role at all soil ESP values and was most likely due to the influence of both excess Na$^+$ and Ca$^{2+}$ deficit at high soil ESP values, where molar concentrations of Mg$^{2+}$ exceeded Ca$^{2+}$.  

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Soil ESP versus Ca/Na ratio for loamy sands was similar to those of the sandy clay samples. The Ca/Na ratio trended toward an expected decrease as soil ESP increased, suggesting that Ca$^{2+}$ was predominantly displaced or exchanged with Na$^{+}$.
however the decrease is most predominant between soil ESP values of approximately 3 – 20.

Clay mineralogy of the studied soils was determined by XRD techniques. The dominant clay present was montmorillonite accompanied by other quartz-based minerals. Appendix 10 contains details of the analysis parameters. Identifying the dominant clay mineral allows for interpretation of Na$^+$-Ca$^{2+}$ selectivity, as discussed in Kopittke et al. (2005). In their study, Na$^+$-Ca$^{2+}$ selectivity for montmorillonite (a 2:1 expanding clay) varied with electrolyte concentration. At small ionic strength and high ESP, the clay platelets were dispersed and dominated by external exchange surfaces, displaying preference for Na$^+$ (Kopittke et al., 2005). However, as clay platelets began to form quasi-crystals (due to increase in ionic strength or decrease in ESP), internal exchange surfaces were created and preference for Ca$^{2+}$ increased (Kopittke et al., 2005). Therefore, the C$_{TU}$ concept is also related to the potential Na$^+$-Ca$^{2+}$ selectivity of a given clay by proxy of dispersion and flocculation.

5.3.6 Sodium loading

The amount of sodium applied to the study site was determined by methods discussed in Section 4.3.3 and is shown in Figure 5.38. Figure 5.38 shows daily sodium loads from effluent irrigation between January 2002 and October 2003. Sodium loading was calculated using sodium values (mg/L) from monthly effluent analysis. Daily sodium loads range from approximately 0 kg/ha to 95 kg/ha (average = 5.3 kg/ha) and were directly related to the volume of STE applied to the 1.32 ha woodlot (Area 3 – Branxton WWTW) (also see Figure 5.2 in this Chapter). As a result, the sodium load applied to Area 3 can be described as uneven throughout the study period.
Due to effluent irrigation and rainfall in January and early February 2002, prolonged surface ponding occurred within the woodlot. As a result, no effluent irrigation occurred during March 2002 as the water was allowed to infiltrate. Irrigation recommenced in April 2002 with the sodium loads applied during April/May 2002 being the largest during the study period. Minimal effluent irrigation occurred in June 2002 due to rainfall and no effluent irrigation occurred in July 2002 due to saturated conditions at many sites ($\theta \gg$ field capacity). The frequency of effluent irrigation became more consistent over the rest of the study period, with the second-largest sodium loads being applied in December 2002 and January 2003. The sodium loads applied are generally reflected in soil ESP results previously presented.
5.3.7 Soil bulk density

Bulk density (BD) was determined by methods described in Section 4.3.4, primarily to determine column-packing specifications for column-leaching experiments. All calculations and sample details are contained in Table 5.8. Results for the first and second bulk density determinations were 1460 kg/m$^3$ and 1550 kg/m$^3$ respectively. These results for soil bulk density at the Branxton WWTW woodlot (Area 3) do not appear to restrict root expansion, where values of $> 1800$ kg/m$^3$ are considered problematic (Jones, 1983). Note that due to the method of sampling for bulk density, only the upper 19 cm is represented in any given sample for bulk density determination.

<table>
<thead>
<tr>
<th>Site</th>
<th>tray mass (g)</th>
<th>wet mass (g)</th>
<th>dry mass (g)</th>
<th>Diff (g)</th>
<th>$\theta$</th>
<th>$\theta$</th>
<th>volume of sample (cm$^3$)</th>
<th>BD (kg/m$^3$)</th>
<th>column radius (cm)</th>
<th>$\pi$</th>
<th>column height (cm)</th>
<th>avg BD (kg/m$^3$)</th>
<th>RSD</th>
</tr>
</thead>
<tbody>
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<td>162.3</td>
<td>7.9</td>
<td>12.1</td>
<td>1346.5</td>
<td>1530</td>
<td>4.75</td>
<td>3.141</td>
<td>19</td>
<td>1550</td>
<td>100</td>
</tr>
<tr>
<td>8</td>
<td>529.7</td>
<td>2242.6</td>
<td>1986.5</td>
<td>256.1</td>
<td>12.9</td>
<td>19.0</td>
<td>1346.5</td>
<td>1480</td>
<td>4.75</td>
<td>3.141</td>
<td>19</td>
<td>1550</td>
<td>100</td>
</tr>
</tbody>
</table>

Table 5.8: Soil bulk density data summary and results

5.3.8 Soil moisture curves

Wilting point (WP) and field capacity (FC) were determined using soil extraction plate methods as discussed in Section 4.3.5. The soil used for this purpose was the combined representative sample taken on the 18/12/01 and 20/2/02. This sample consisted of all less than 20 cm samples being placed together (from S1, S2, S3, S4, S5, S6, S7, S8, S10, S11, S12 and S13). Combining of depth-equivalent samples continued,
resulting in representative soil samples for the less than 20 cm (equal masses of the 0-10 cm and 10-20 cm fractions), 40 – 60 cm and 80 – 100 cm in Area 3. Figures 5.37 to 5.39 show the results from these tests. The 95 % confidence interval is shown by error bars, while plant available water (PAW), FC and WP are also highlighted.

Figure 5.39 shows the soil moisture curve for the less than 20 cm representative woodlot sample, highlighting a FC of approximately 41 %, a WP of approximately 6.5 % and resultant PAW being 34.5 %. Figure 5.40 shows the soil moisture curve for the 40 – 60 cm representative woodlot sample, highlighting a FC of approximately 32 %, a WP of approximately 7 % and the resultant PAW being 25 %. Figure 5.41 shows the soil moisture curve for the 60 – 80 cm representative woodlot sample, highlighting a FC of approximately 34 %, a WP of approximately 7.5 % and the resultant PAW being 26.5 %.

Compared to $\theta$ obtained in the field at each site and depth, the irrigated sites S1 and S2 were particularly dry at depths greater than 10 cm. In contrast, the irrigated sites S4, S5, S6 and S10 were saturated or near-saturation at depths greater than 40 cm, although these sites all had sandy clay present at depth.

![Figure 5.39: Soil moisture curve for the representative sample (0 - 20 cm).](image)
Figure 5.40: Soil moisture curve for the representative sample (40 - 60 cm)

Figure 5.41: Soil moisture curve for the representative sample (80 - 100 cm)
The soil profile at the non-irrigated site S9 was saturated at depths of greater than 60 cm, from February 2002 to October 2003. This is unusual in that S9 never received irrigation. S14 remained relatively dry, with $\theta$ values rarely approaching 20%.

5.4 Column-leaching experiments

5.4.1 Column-leaching in relation to the $C_{TU}$ and $C_{TH}$

The aim of this experiment was to observe the effect on optimum permeability when either STE or rainwater are applied to effluent irrigated woodlot soils and highlight the implications for soil management and irrigation scheduling. The presence of dispersed clay particles in the output solution after equal volumes of STE and rainwater had been applied would indicate the extent of micro-aggregate/soil pore stability induced by the measured range in the SAR of the applied waters. This phenomenon was difficult to observe in the field because of the high hydraulic conductivity of the loamy sands present, however the physiochemical response has been validated by Quirk and Schofield (1955) and Davidson and Quirk (1959).

Figure 5.42 shows the results of applying the same soil (soil ESP = 21.6) with a solution that has an SAR $> C_{TU} < C_{TH}$ (Column 1) and a solution with an SAR $< C_{TU}$ (Column 2). The Column 1 output sample was relatively clear and indicated that soil structure was maintained ($> C_{TU}$). Approximately 98% of the applied 500 mL permeated through the column after approximately 12 hours and no ponded effluent remained on the surface.

The Column 2 output sample was turbid and the eluted soil (rainwater) was observed to be less permeable than Column 1 (eluted with STE). After 12 hours, the remaining rainwater in the volumetric flask (approximately 100 mL) in Column 2 was removed from the experiment. Column 2 still had approximately 1 cm of rainwater yet to permeate. This took a further 8 hours to permeate into the soil and cease output from Column 2.
Figure 5.42: Column-leaching in relation to the $C_{\text{TU}}$ and $C_{\text{TH}}$, showing a stable irrigated soil (column 1, SAR of application waters >$C_{\text{TH}}$) and an irrigated soil where dispersion has taken place (column 1, SAR of application waters <$C_{\text{TU}}$).

Figure 5.42 was taken approximately 24 hours after the rainwater had permeated into the soil, that is, approximately 44 hours after the start of the experiment. Only 70% of the 500 mL of rainwater permeated through Column 2 over the same time, which relates to a permeability decrease of approximately 25%. The gravimetric soil moisture ($\theta_g$) was determined to be approximately 25% for both columns. Multiplied by the average bulk density from field results, this equates to a $\theta$ of approximately 38%. This is comparable to results obtained for field capacity in the soil moisture curves shown in Section 4.3.5.
5.4.2 Column-leaching by depth

The purpose of these experiments was to determine relative changes in permeability with respect to hydraulic loading, the pH, EC, SAR of the application waters and the ESP of the receiving soil. Since dispersed clay particles resulting from micro-aggregate instability appear when the SAR of the application waters is less than the soil ESP (as a C_{TU}), the turbidity of the eluent was also measured. In addition, solute breakthrough curves are shown that validate the rapid rate of sodium transport through the soil profile, at different depths.

Soil from each depth increment was undertaken in triplicate, as discussed in Section 4.3.6, although only the average values of each series are shown in the following Figures. Each depth increment will be presented together in the same Figure. All column-leaching results can be found in Appendices 8 and 9.

Eight solution additions were made to each of the soil columns: 100 mL of rainwater (0 minutes), 250 mL of STE, 250 mL of STE, 100 mL of rainwater (210 minutes), 250 mL of STE, 50 mL of rainwater (450 minutes), 250 mL of STE and 100 mL of rainwater (570 minutes). The timing of sample collection and the volume received are shown in Appendices 8 and 9. These represented excessive events for both STE irrigation and rainfall. In the columns, 250 mL of STE was equivalent to an irrigation event of approximately 133 mm, 100 mL of rainwater was equivalent to approximately 53 mm of rainfall and 50 mL of rainwater approximately equal to 27 mm of rainfall. Rainwater was applied intermittently as to mimic STE irrigation interrupted by rainfall and to observe and measure the change in chemistry of the output samples.

Each addition was applied when the previous addition had permeated into the soil. It must be noted that the first output sample from the 10 – 20 cm increment was introduced into the 20 – 40 cm columns and resulted in zero permeation after 12 days. The experiment ended at this point. Figures 5.40 A – E summarise the 0 – 10 cm increment and 10 – 20 cm increments, showing variations in pH, EC, SAR, permeability and turbidity, respectively, over time. Figure 5.43F shows the solute breakthrough curves (C/C_{o}) for both series.
Table 5.9 shows the input and output chemical parameters for the STE and rainwater used during this experiment. Table 5.10 shows the input soil chemistry and output results from soil analyses after column-leaching.

**Table 5.9: Chemistry of waters used for CLE.**

<table>
<thead>
<tr>
<th></th>
<th>µS/cm</th>
<th>NTU</th>
<th>meq/L</th>
<th>pH</th>
<th>EC</th>
<th>Turb.</th>
<th>Ca²⁺</th>
<th>Mg²⁺</th>
<th>Na⁺</th>
<th>K⁺</th>
<th>SAR</th>
</tr>
</thead>
<tbody>
<tr>
<td>Rainwater</td>
<td>5.74</td>
<td>100</td>
<td>1</td>
<td>0.144</td>
<td>0.030</td>
<td>0.190</td>
<td>0.010</td>
<td>0.5</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>STE</td>
<td>6.65</td>
<td>780</td>
<td>1</td>
<td>0.976</td>
<td>0.252</td>
<td>4.137</td>
<td>1.015</td>
<td>5.2</td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Table 5.10: Solute exchange medium chemical parameters as used for CLE, showing before/after results of each depth requirement.**

<table>
<thead>
<tr>
<th>Soil Chem (before)</th>
<th>1:5 pHw</th>
<th>1:5 (dS/m)</th>
<th>EC</th>
<th>1M NH₄Cl (cmol(+)/kg)</th>
<th>cmol(+)/kg</th>
<th>% CEC</th>
<th>ESP</th>
</tr>
</thead>
<tbody>
<tr>
<td>0-10</td>
<td>5.59</td>
<td>0.015</td>
<td>0.908</td>
<td>0.286</td>
<td>0.328</td>
<td>0</td>
<td>1.522</td>
</tr>
<tr>
<td>10-20</td>
<td>5.61</td>
<td>0.011</td>
<td>0.53</td>
<td>0.286</td>
<td>0.305</td>
<td>0</td>
<td>1.121</td>
</tr>
</tbody>
</table>

(after)

| Column A          |         |     |     |     |     |     |     |     |
| 0-10              | 6.39    | 0.02 | 0.926 | 0.239 | 0.109 | 0.07 | 1.344 | 8.1  |
| 10-20             | 6.19    | 0.032 | 0.813 | 0.242 | 0.169 | 0.05 | 1.274 | 13.3 |

(after)

| Column B         |         |     |     |     |     |     |     |     |
| 0-10             | 6.40    | 0.02 | 0.912 | 0.247 | 0.078 | 0.049 | 1.286 | 6.1   |
| 10-20            | 5.88    | 0.031 | 0.833 | 0.329 | 0.178 | 0.05 | 1.39  | 12.8  |

(after)

| Column C         |         |     |     |     |     |     |     |     |
| 0-10             | 6.70    | 0.024 | 0.864 | 0.242 | 0.093 | 0.033 | 1.232 | 7.5   |
| 10-20            | 5.77    | 0.029 | 0.781 | 0.324 | 0.142 | 0.02 | 1.267 | 11.2  |

AVGE of all 3 columns

| 0-10             | 6.50    | 0.021 | 0.901 | 0.243 | 0.093 | 0.051 | 1.287 | 7.2   |
| 10-20            | 5.95    | 0.031 | 0.809 | 0.298 | 0.163 | 0.040 | 1.310 | 12.4  |

(after – before = difference)

| 0-10             | 0.907   | 0.006 | -0.007 | -0.043 | -0.235 | 0.051 | -0.235 | -14.3 |
| 10-20            | 0.337   | 0.020 | 0.279  | 0.012  | -0.142 | 0.040 | 0.189  | -14.8 |

Table 5.10 shows that the 0 – 10 cm series increased in pH by 0.91, the EC increased by 0.006 dS/m and exchangeable K⁺ increased by 0.051 cmol(+)/kg. However, exchangeable Ca²⁺, Mg²⁺ and Na⁺ decreased by 0.007, 0.043 and 0.235 cmol(+)/kg respectively. The ESP fell by approximately 14.3%.

Table 5.10 shows that the 10 – 20 cm series increased in pH by approximately 0.34, the EC increased by 0.020 dS/m, exchangeable Ca²⁺ increased by 0.279
cmol(+)/kg, exchangeable Mg$^{2+}$ increased by 0.012 cmol(+) kg and exchangeable K$^+$ increased by 0.040 cmol(+) kg. However, exchangeable Na$^+$ decreased by 0.142 cmol(+) kg and the ESP fell by approximately 14.8%. Cation exchange capacity (CEC) decreased during both the 0 – 10 cm and 10 – 20 cm series, showing leaching of cations from column to column.

Figure 5.43A shows variation in pH over time for the 0 – 10 cm and 10 – 20 cm series. Rainwater with a pH of 5.74 was first added to each column. For the 0 – 10 cm series, Figure 5.43A shows that the pH of the first output sample had increased (~ 6.6). Upon addition of STE, the pH declined until the second addition of rainwater at approximately 210 minutes. The pH peaked sharply at 240 minutes, before declining to a minimum (~ 5.1) at 400 minutes. The pH rises at 450 minutes, coinciding with the addition of 50 mL of rainwater and peaks at 5.40. After the last STE addition, the pH declined to a minimum (~ 5.2) at 550 minutes, which then rose with the last addition of rainwater to approximately 5.40 at 570 minutes. Successive peaks observed in Figure 5.43A and caused by the addition of rainwater, decrease in height over time. In the 10 – 20 cm series similar trends were observed, although variations are less in magnitude. Also, the 10 – 20 cm series appears out of phase with the 0 – 10 cm series, albeit slightly, and is due to different sample times during the experiment period.

Figure 5.43B shows variation in EC (µS/cm) over time for the 0 – 10 cm and 10 – 20 cm series. Rainwater with an EC of 100 µS/cm was first added to each column. Trends in Figure 5.43B are almost the reverse to those in Figure 5.43A. Rainwater, having a lower EC than STE, caused EC to decrease when output through each column. As a result, the first output sample had an EC of 300 µS/cm. After addition of STE, the EC increased to approximately 700 µS/cm. Addition of rainwater at 240, 450 and 550 minutes caused the EC to decrease to a minimum on every occasion (range of minima = 100 – 380 µS/cm). However, the addition of STE at 100, 300 and 510 minutes caused the EC of the eluted solution to increase to a maximum on every occasion (range of maxima = 650 – 680 µS/cm). The 10 - 20 cm series again appears out of phase with 0 - 10 cm series.
Figure 5.43: A – variations in pH; B – variations in EC (µS/cm); C – variations in SAR over time in column leaching experiments

STE = secondary treated effluent
R = rainwater
Figure 5.43: D – variations in permeability; E – variations in turbidity (NTU); F – C/Co curves over time in column leaching experiments.
Figure 5.43C shows variation in SAR of the output sample over time for the 0 – 10 cm and 10 – 20 cm series. Rainwater with an SAR of 0.5 was first added to each column. For the 0 – 10 cm series, the first output sample had an SAR of 1.7, which had increased to 2.8 after the first STE addition (100 minutes). Addition of rainwater at 240, 450 and 550 minutes caused the SAR to decrease to a minimum on every occasion (range of minima = 1.1 – 2.2). However, the addition of STE at 100, 300 and 510 minutes caused the SAR to increase to a maximum on every occasion (range of maxima = 2.8 – 3.0). This has ramifications for dispersion events and will be further discussed in Chapter 6.

For the 10 – 20 cm series, the first output sample had an SAR of 1.6, which had increased to 4.8 by 100 minutes. SAR values increased for every output sample after 100 minutes, with a maximum of 8.5 at 510 minutes. When compared to Figure 5.43B, the 10 – 20 cm series shows poor correlation between SAR and EC ($r^2 = 0.16$, see Appendix 10). Conversely, the 0 – 10 cm series shows a strong correlation between SAR and EC ($r^2 = 0.94$, see Appendix 9). This highlights the difficulty in determining SAR and/or EC at any soil depth < 10 cm.

Figure 5.43D shows variation in permeability of the output sample over time for the 0 – 10 cm and 10 – 20 cm series determined using Equation 20 (p73). Permeability decreased, ranging between approximately 55 and 140 mm/hr for the 0 –10 cm series and 35 to 90 mm/hr for the 10 – 20 cm series. A cyclic variation in permeability from minimum to maximum occurs approximately every 100 minutes due to the water additions. Since disturbed soil columns were used and overall $K_{sat}$ was relatively high, the large variation of $K_{sat}$ were assumed to be due to the alternating falling head - constant head conditions that existed during the experiment and not the timing and/or electrolyte effects of the added waters. However, translocated clay particles indicate that blocked soil pores may have impacted on the observed variation in $K_{sat}$ although the extent cannot be determined from the data presented.

It must be noted that if accurate hydraulic conductivity measurements of in-situ soils are to be determined, then soil cores sampled in-situ should be used. However, these experiments were conducted to show the difference in permeability rates due to the effects of increasing Na$^+$. Therefore, whilst the comparison of hydraulic conductivity is
not a true reflection of in-situ soil hydraulic conductivity, the experiment did encourage saturation of all soil pores and observation of reduction in pore size due to the variation in cationic charge driven by variation in permeant chemistry (volume-weighted average SAR); and the potential for dispersion and subsequent translocation of clay particles and/or blocking of deformed soil pores (micro-aggregate/soil pore stability). This phenomenon governs the deviation from optimum permeability for a given soil chemistry and effluent irrigation scheduling condition.

Figure 5.43E shows variation in turbidity (NTU) of the output sample over time for the 0 – 10 cm and 10 – 20 cm series. Rainwater with a turbidity of 1 was first added to each column. For the 0 – 10 cm series, the first output sample had a turbidity of approximately 200 NTU, which had decreased to 1 after the first STE addition (100 minutes). Addition of rainwater at 240 minutes caused the turbidity to increase to approximately 40, and 9 at 450 minutes. Negligible increase was observed at 550 minutes. However, the addition of STE at 100, 300 and 510 minutes caused the turbidity to decrease to a minimum on every occasion (minimum turbidity range = 0.9 – 1.5 NTU).

For the 10 – 20 cm series, the first output sample had a turbidity of 30 NTU, which had decreased to 6 at 100 minutes. Turbidity values ranged between 1 and 10 NTU for the rest of the experiment. This shows that micro-aggregate/soil pore stability comes closer to equilibrium with the permeating waters as it moves deeper in the soil profile. When compared to Figure 5.43B, the 0 – 10 cm series shows a negative correlation between turbidity and SAR ($r^2 = 0.46$, see Appendix 9). Similarly, the 10 – 20 cm series shows a similar negative correlation between turbidity and SAR ($r^2 = 0.51$, see Appendix 10).

Figure 5.43F shows the C/C_o versus time (solute breakthrough curves) for the 0 – 10 cm and 10 – 20 cm series. The 0 – 10 cm series was plotted as discussed in Section 4.3.6, using the sodium concentration of the STE as C_o and the sodium concentration of the output sample as C. The 10 – 20 cm series was plotted using the sodium concentrations of the 0 - 10 cm series output sample as C_o and the sodium concentrations of the 10 - 20 cm series output sample as C. Figure 5.43F must be viewed as a series of solute break-through curves. For example, input samples have a range of SAR values
over the experiment duration (580 min). As a result, the solute break-through curves are not typical of the example given in Chapter 4.3.6. Therefore, trends in the solute break-through curves must be interpreted with this in mind.

The first output sample in the 0 – 10 cm series had a $C/C_0$ value of 0.5, which had increased to 1 by 125 minutes. Therefore, the ease of transmission for sodium is validated as being rapid through the soil profile (low retardation factor, $R_d$, from Equation 7). When the $C/C_0$ equals 0.5, the solute is said to “break-through” and the time taken to do this determines the dominant solute process operating (advection and/or hydrodynamic dispersion). Branxton WWTW woodlot soils show a rapid increase to $C_o$, thus advection appears to be the dominant process.

Addition of rainwater at 0, 210, 450 and 570 minutes caused the $C/C_0$ to fall to ≤ 0.6. Due to the decreased SAR, clay minerals swell as rainwater permeates through the column and $C_o$ of the permeating solution attempts to attain $C$. This is adequately explained by the fact that input solutions had a range of SAR values, of which $C$ from the first output sample had the minimum SAR value.

The first output sample in the 10 – 20 cm series had a $C/C_0$ value of 0.7, which had increased to 2.1 by 125 minutes. The 10 - 20 cm series experiences less uniform increases and decreases in $C/C_0$, which are due to variations in the SAR and dispersed clay contents (as turbidity) of the output sample from 0 - 10 cm series and being introduced to the 10 – 20 cm columns. The solute break-through curve for the 10 – 20 cm columns had to be calculated using initial concentrations from 0 – 10 cm input waters, therefore, 10 – 20 cm solute break-through curves appear to be “piggy-backed” on the 0 – 10 cm curve. However, $C/C_o$ values for the 10 – 20 cm curve peak and fall as for the 0 - 10 cm curve and results can be interpreted in a similar manner for both.

5.5 The Soil ESP/Effluent SAR continuum for micro-aggregate/soil pore stability

Many of Quirk and Schofield’s results from 1955 have still yet to be fully appreciated and in particular, the significance of the $C_{TU}$ (Quirk, 2001). In this thesis, an attempt has been made to incorporate years of soil research into a management tool that
can be applied to any woodlot soil irrigated with STE. This continuum, which is presented in Figure 5.44, predicts the relative stability of micro-aggregates/soil pores under the influence of irrigation waters of a known SAR and for a receiving soil of any texture and known ESP value. The degree of stability of micro-aggregates/soil pores within the soil profile is determined from the Threshold concentration (C_{TH}) and Turbidity concentration (C_{TU}) calculations (Equations 4 and 5 respectively).

The continuum allows for any soil of known ESP to be assessed for potential dispersion under irrigation waters of known SAR. The work by Cass and Sumner (1982) is also expanded on with this continuum, as the C_{TU} has been incorporated to provide a second “point” on the sodium stability curve.

In Chapter 6.4.3, a hypothetical application of the continuum will be discussed. This highlights the significance of understanding sodium flux in a woodlot soil receiving STE, and in particular, understanding the probable impact on micro-aggregate/soil pore stability under STE irrigation schemes to woodlots.

The soil ESP/effluent SAR continuum for micro-aggregate/soil pore stability emerged as a product of 50 years of soil research. This continuum has been recognised in the past (Crescimanno et al., 1995; Meneer et al., 2001), but has never been applied to field conditions. Also, past researchers had developed a misconception in the literature, as discussed in the work of Quirk (2001). Previous research used the C_{TH} as the critical electrolyte concentration for soil stability, and as a result, missed the importance of dispersion in permeability measurements (Quirk 2001).

This misconception highlights how soil research in the past has realised the importance of electrolyte concentration on maintaining soil structure, although little acknowledgement has been given to the Turbidty concentration (C_{TU}). The C_{TU} defines the critical electrolyte concentration at which micro-aggregates begin to disperse, translocate to lower depths and block soil pore continuity, thus reducing permeability. In light of this misconception, this study shows that temporal sodium flux in an effluent irrigated woodlot does influence optimum permeability. Using the C_{TH} and C_{TU} as proxy indicators for micro-aggregate/soil pore stability allows adequate assessment of optimum permeability with respect to soil ESP/effluent SAR over time.
Figure 5.44: Soil ESP/Effluent SAR continuum for micro-aggregate/soil pore stability.

Adapted from several sources, namely Quirk and Schofield (1955); Rensgamy et al (1984); Quirk (2001) and Shainberg et al (2001).
The holistic approach taken during this thesis allowed various components of an effluent irrigated woodlot to be monitored and assessed in relation to temporal sodium flux. Comparison of $\Delta$ESP values with cumulative irrigation surplus/deficit, $\theta$, the volume-weighted average SAR of the application waters, and groundwater depth over time provided suitable proxy-indicators for sodium flux and optimum permeability by use of the continuum. These indicators allow irrigation managers to not only satisfy irrigation scheduling issues, but also predict optimum permeability and dispersive conditions over time if adequate monitoring steps are observed.

Due to the work of Norrish and Quirk (1954), Quirk and Schofield (1955), McNeal and Coleman (1966), Cass and Sumner (1982), Rengasamy et al., (1984) and Shainberg et al., (2001), optimum soil permeability occurs when the permeating waters have an SAR that lies between the $C_{TU}$ and the $C_{TH}$. This is particularly relevant for the uppermost 10 cm of the soil receiving STE, as this is the boundary layer most susceptible to loss of soil structure from rainwater of low SAR as soil ESP increases. The use of both the $C_{TU}$ and $C_{TH}$ is a relatively new concept that addresses the misconception highlighted by Quirk (2001).

Therefore, results from permeability tests will be dependent on both the soil ESP and the effluent SAR of the solution used in the permeability test. This should be borne in mind when interpreting permeability data. For example, consider a soil of ESP = 12, where the $C_{TU} = 2.1$ and the $C_{TH} = 7.3$. For optimum permeability to occur, the SAR of the application waters should lie between 2.1 and 7.3. But many permeability tests use distilled water or tap water with SAR values less than 2. Thus, permeability tests using application waters less than SAR 2 would cause significantly lower results.

An improved approach would be to create a series of permeability reduction versus ESP curves for a given soil, whereby the soils’ response to a range of effluent SAR values could be visualised using the continuum, similar to Figure 2.6 (Chapter 2). The extremes in response of the Branxton woodlot soil used in this study, after being applied with STE and rainwater and effluent of (i) $>>C_{TU}$ but $<C_{TH}$ and (ii) $<<C_{TU}$ respectively, are shown in Figure 5.42. Therefore, the soils’ response can be determined using the continuum, which acts as a management tool for scheduling soil and/or...
effluent amendment. In this way, the soil ESP/effluent SAR continuum for microaggregate/soil pore stability predicts the probable response of a soil of known ESP irrigated with an effluent of known SAR.

Since sodium is the dominant electrolyte in secondary treated effluent, using the SAR of the applied waters instead of total electrolyte concentration is more useful in conceptualising the impact sodium flux has on the response of the woodlot soil to the applied waters. Note that according to Equation 2, the SAR can be decreased by addition of calcium and/or magnesium. Of these, calcium is the preferred dominant electrolyte in a healthy soil system (McBride, 1994). In an effluent irrigated woodlot it is desirable to maintain soil “health” and to do this, a variety of management strategies exist.

In the field, the electrolyte concentration range applied to the woodlot at Branxton was approximately 0.5 – 8.1 meq/L, including the effect of rainfall chemistry. The range of volume-weighted average SAR values ranged from approximately 0.5 – 5.9 for the same waters (see Figure 5.21, Section 5.3.1). The relationship between electrolyte concentration and SAR for this study is shown in Figure 5.45, and validates the use of average volume-weighted SAR values as a surrogate for electrolyte concentration ($r^2 = 0.89$) in the continuum.

Figure 5.45: Effluent/rainfall SAR versus electrolyte concentration
5.6 Summary

Results from monitoring and laboratory analyses have been presented in this chapter. Results from monitored sodium loading, soil moisture curves and column-leaching experiment results have also been shown. Trends in several water balance components for a woodlot soil irrigated with STE have been investigated in this Chapter (Aim 1). The temporal and spatial variations, in both the water balance components and measured soil properties, particularly the sodium flux, have been determined (Aim 2). The most significant results and trends to be noted include:

- The hydraulic loading to Area 3 (Figure 5.2), potential evapotranspiration (PET) (Figure 5.2), daily irrigation surplus/deficit (Figure 5.4) and the cumulative irrigation surplus/deficit (Figure 5.5);
- Variations in groundwater depth, indicating the response of groundwater levels to effluent irrigation (Figures 5.21A - D and 5.5);
- Weekly volumetric soil moisture ($\theta$) averaged from all monitored sites within the woodlot (Figure 5.20) showing the net weekly change in $\theta$ for all sites;
- Average ESP and average $\Delta$ESP for the study period, which highlight the dynamic nature of soil sodium at irrigated and non-irrigated sites (Figure 5.30), as well as individual site comparisons, specifically S1 (irrigated) versus S14 (non-irrigated) and S10 (irrigated) versus S9 (non-irrigated).
- Soil moisture curves, particularly in reference to FC and WP (Figures 5.37 to 5.39);
- Results from column leaching experiments indicating the displacement of sodium and clay particles from the surface 10 cm, through to the 10 – 20 cm and implications for further percolation (Figures 5.41A – F);
- The soil ESP/Effluent SAR continuum for micro-aggregate/soil pore stability (Figure 5.44).

The implications of the sodium flux on the loss of soil structure, in relation to soil ESP and effluent SAR, which have been investigated (Aim 3), highlight the potential for loss in soil structure under specific conditions.
Chapter 6: DISCUSSION

6.1 Introduction

The ability to define the sodium flux over time depends on the frequency of soil sampling and the ability to interpret the net loss/gain in soil sodium in relation to the applied hydraulic load. Past research has measured soil ESP on an annual basis, or longer, making it impossible to interpret temporal sodium flux. This thesis has reported and discusses results from a two-year monitoring regime of the temporal sodium flux in a woodlot soil receiving secondary treated effluent at Branxton, NSW.

The net loss/gain of exchangeable sodium (as $\Delta$ESP) at different depths and times was compared with the cumulative irrigation surplus/deficit over time. Volumetric soil moisture ($\theta$) was also measured on a weekly basis. Groundwater pH, EC and SAR were monitored, although discussion during this chapter will focus on changes in groundwater depth.

The need to extrapolate laboratory based theory, regarding soil micro-aggregate stability, to field site conditions has been highlighted in past research (Sumner, 1993; Qadir et al., 2000; Oster and Shainberg, 2001; Qadir and Schubert, 2002). This study aimed to investigate temporal sodium flux in a woodlot soil receiving STE and now discusses this flux in relation to soil structure and the implication for soil management and STE irrigation scheduling. For example, the monitoring programs initiated represent proxy-indicators that could be used to monitor likely trends in the sodium flux during effluent irrigation to woodlots, the results of which were presented during Chapter 5.

In Section 5.5 the impact of increasing sodium on soil structure was graphically described as a continuum which showed the potential impact of soil ESP relative to the effluent electrolyte concentration of applied waters and rainfall (as a volume-weighted average SAR) on optimum soil permeability. This should be considered when assessing irrigation scheduling and in making soil management decisions. However, caution must be shown as results also showed that soil ESP could be misleading in the long-term, due to variations in $Ca^{2+}$ and $Mg^{2+}$ disguising the actual $Na^+$ variation in the soil profile, particularly during non steady-state percolation.
Results from monitoring and laboratory experiments are discussed in terms of how they fit into the existing body of knowledge and whether they give new insights into managing sodium in soils of STE irrigated woodlots. The ability to implement adaptive strategies on a dynamic system is also discussed. Relevant outcomes and future research direction are then summarised during Chapter 7.

6.2 Effluent Irrigation Scheduling

In order to not disrupt the water balance in an effluent irrigated woodlot, it is necessary that a reuse scheme be designed to assimilate the applied water (and nutrients). Over-irrigation can induce surface runoff and/or waterlogging while under-irrigation causes stress to the trees (Snow et al., 1999). Therefore, the effluent application rate must be closely related to the water used by the trees on a temporal basis (Potential evapotranspiration (PET)).

From this study, maintaining sufficient soil water in the root zone to promote sodium leaching has been shown to be a primary concern. Each component of the water balance was described in Section 2.3.3 (Equations 10 and 11). The water balance component results described and presented showed temporal trends, which are used to highlight sodium flux and potential soil response due to effluent irrigation and rainfall. Assumptions and limitations of irrigation scheduling calculations will now be discussed.

Potential evapotranspiration (PET) was determined by methods discussed in Section 4.2.3. PET represents the evapotranspiration component of the water balance and while lacking site-specific Penman-Monteith complexity (detailed in Raupach, 1995), does give temporal trends in potential evapotranspiration. The limitation of using class A pan data and pan co-efficient \( K_p \) is that they represent only a general guide to true site-specific evapotranspiration rates. However, PET will differ across the woodlot depending on the size and health of individual trees and the extent of understorey growth and canopy closure (Myers et al., 1999). Daily PET was shown in Figure 5.3 (Section 5.2.3) and indicated that site PET values are typically at a maximum during the summer months and at a minimum during winter months. The fact that high daily PET conditions occurred during the winter months emphasised the high degree of daily management required to maintain sufficient soil water for woodlot growth and managing soil sodium.
Effluent (Qe) and rainfall (Qp) loading were determined using methods discussed in Section 4.2.2. Daily results for the study period were shown in Figure 5.2 (Section 5.2.2). One assumption made was that both effluent and rainfall were evenly distributed over Area 3, although in reality, this would not be the case. Qe was applied by use of a sprinkler system, which has been shown to be of limited efficiency (Myers et al., 1999). Myers et al. (1999) use an irrigation efficiency value of 0.7 for the WATSKED model, as losses occur due to evaporation and spray mist moving off-site. No irrigation efficiency value was used in this study, although interception loss (IL) was used due to the numerous grasses, weeds and other understorey vegetation in the woodlot that would have prevented a percentage of the applied effluent and rainfall reaching the ground.

Interception loss (IL) is the capture and storage of rainfall and applied effluent by vegetation within the woodlot and was represented as a percentage of the total hydraulic load in water balance calculations (Equation 11). IL was incorporated into water balance calculations using methods discussed in Section 4.2.4. Two values were used, a spring/summer value (9%) and an autumn/winter value (6%). Each of these was assumed to be constant throughout their given seasons, although in reality, would have varied depending on various factors (Battaglia and Sands, 1997).

These factors include the density of undergrowth, the intensity and frequency of rainfall and the canopy cover of the woodlot. For example, if rainfall is less than 5 mm/day, IL would be close to 100% due to the light rains being absorbed or captured, which prevents water reaching the ground (Myers et al., 1999). In contrast, during heavy rainfall, IL could be as low as 5-10%. For a flooded gum (E. grandis) plantation at Wagga Wagga, IL was found to be 12% in winter and 18% in summer (Myers et al., 1999). In the study by Myers et al. (1999), IL was determined by using typical values from past research (Myers et al., 1996).

Other models use an IL value based on leaf-area index calculations (Snow et al., 1999). In any case, and acknowledging that some IL occurs, the ability to determine accurate and precise IL values in the field was outside the scope and resources of this study. Due to this, volumetric soil moisture gave the best indicator of the net volume of...
applied effluent and/or rainfall reaching the soil, highlighting the importance of monitoring volumetric soil moisture in effluent irrigation schemes.

To prevent excessive drainage losses to groundwater, runoff (R) and deep drainage (D) were designed to be zero, although in reality, this proved not to be the case. Under this assumption, the aim of using Equation 11 was to create a quantitative indicator of the daily irrigation deficit/surplus over the study period (Figure 5.4, Chapter 5). The cumulative irrigation surplus/deficit (Figure 5.5, Section 5.2.3) resulted in the identification of longer-term wetting and drying periods of the soil profile. From Figure 5.5, wetting periods included the first week of February 2002, April – June 2002, December 2002 – mid January 2003 and mid February – June 2003. Drying periods included mid February – April 2002, June – December 2002, mid January – mid February 2003 and June – October 2003.

Ultimately, it was believed that the approach used both over and underestimated cumulative irrigation surplus/deficit due to:

- The overestimation of Kp;
- The underestimation of actual surface runoff and;
- The overestimation of actual effluent loading due to potential irrigation distribution issues.

In hindsight, the use of Kp values greater than 1 was probably not the appropriate assumption, however the use of a daily time-step for PET, rather than a weekly time-step as used in WATSKED (Myers et al., 1999) gave greater resolution of weekly variation.

Actual surface runoff was not observed because when significant rainfall events of the highest intensity occurred, no person was present on site at the time. Observations were made the following day after high rainfall events were recorded.

Effluent irrigation was assumed to distribute 100 % to the spray-irrigated Area 3 (the studied woodlot) for a given amount of time, then 100 % to the drip-irrigated Area 2 at a different time. Towards the end of the study period, plant operators revealed a discrepancy in the actual flow rates, that is, for much of the study period approximately 40 % of the “assumed” volume may have been applied to Area 2 during the same scheduled application to Area 3. This means there was a potential overestimation in the
The actual volume of effluent that was applied to Area 3 at various times between January 2002 and October 2003.

The direction of sodium flux (accumulation and/or leaching) was indicated from the slope of the cumulative irrigation surplus/deficit, relative volumetric soil moisture and the subsequent response of other monitored parameters. For example, an increasing cumulative irrigation surplus was attributed to excessive hydraulic loading, while a decreasing cumulative irrigation deficit was attributed to excessive PET (under-irrigated), although both scenarios could have similar volumetric soil moisture at any given time. As such, a “true” water balance could not be ascertained solely from monitored water balance components. However, the amount of sodium added through effluent irrigation, particularly when effluent irrigation depths exceed 20 mm, does influence the ESP of woodlot soils. Interaction between sodium loading, cumulative irrigation surplus/deficit, $\Delta$ESP, change in sodium (cmol(+)/kg) and $\theta$ are further discussed using specific site comparisons in Section 6.3.1.

6.2.1 Effluent and rainfall chemistry

Results from Table 5.3 (section 5.3.1) showed that sodium is by far the dominant cation contained in the STE. The chemical composition of STE at the Branxton WWTW was within the range of other secondary treated wastewaters emanating from wastewater treatment works throughout New South Wales (Patterson, 1994), with respect to the parameters measured. From June 2002 to October 2003, variations in sodium concentrations altered the SAR by approximately 5 units, resulting in an average effluent SAR of 4.7 (RSD = 0.9, confidence interval 95 %). The impact of the temporal variation in effluent SAR on soil ESP and soil structure is further discussed in Sections 6.4.

6.2.2 Groundwater

Site groundwaters have been shown to be highly variable in EC and salinity, and that they exist within 0 and 2.5 m from the surface. GW2 was selected as a proxy-indicator for whether the cumulative irrigation surplus/deficit was increasing, declining or stable, due to its rapid and varied response to hydraulic loading. In addition, GW2 exists at the lower corner of the woodlot, and with GW5, groundwater levels were nearer
the surface than other sites, for example, groundwater was at a depth of 0.5 m at GW2 and nearer to 2.5 m at GW3.

Groundwater moves through a combination of sandstone bedrock and the overlying sandy clay according to a previous geo-technical report (HWC, 1987). The bores themselves did not penetrate into the sandstone bedrock, thus the groundwater rate determined was most applicable for the sandy clay encountered at depth in the woodlot studied. The groundwater flow rate of 0.9 mm/h was much less than that expected for sandy clay, thus caution must be shown in interpreting the result.

Loamy sand generally has a greater permeability than sandy clay. In the case of Area 3, if irrigation and/or rainfall saturated the loamy sand A horizon, further irrigation must not exceed the permeability rate of the sandy clay B horizon. When a surface slope is greater than 3 %, lateral movement of soil water may occur, thus giving an overestimated groundwater flow rate. Lateral soil water movement occurs when permeating water reaches a rate limiting boundary within the soil profile, as in a clay layer of lesser permeability, and starts to move down slope within the A horizon.

Whether increases in EC reflect solute “pulses” to groundwater is unclear with the data presented. However, increased EC at all sites occurred approximately three months after significant increases in cumulative irrigation surplus/deficit, particularly after rainfall. For example, the June 2002 increase in cumulative irrigation surplus/deficit could be reflected in the September-October-November 2003 EC increases, or the June 2003 increase in cumulative irrigation surplus/deficit could be reflected in the October-November 2003 EC increase.

No interpretation of solute losses was made in this study, as groundwater levels were only used as an indicator of whether applied the depth of irrigation applied directly influenced groundwater levels. No individual event or sequence of irrigation events, were observed to increase groundwater levels in this study.

**6.2.3 Volumetric soil moisture (\(\theta\))**

Discussion will focus on the average of all sites (Figure 5.20), although sites S9 and S14 (Figure 5.15 and 5.19 respectively) will be discussed as non-irrigated sites, which are representative of the profiles shown in Figure 3.3 (Chapter 3).
Volumetric soil moisture acts as an indicator of whether the soil is saturated and susceptible to changes in R and D (from Equation 10, chapter 2), although solute transport would be assumed to be at a maximum for a solute of low retardation factor ($R_d$) in a saturated soil profile. From column-leaching experiments, this was found to be the case for sodium (refer Section 6.4.1). When non-saturated, solute transport is limited by the depth of effluent and/or rainfall received (Figure 2.11, Chapter 2). Under non-saturated conditions, the probability of R and D being zero increases, due to sufficient soil water storage capacity in accepting the hydraulic load. Micro-aggregate/soil pore stability will ultimately be dictated by the continuum, however, even at optimum permeability, sodium would be assumed to be accumulating in the root zone (<1 m) under non-saturated conditions.

S9 and S14 did not receive irrigated effluent. S9 had sandy clay at a depth greater than 40 cm, which remained relatively moist for the entire monitoring period ($\theta > 35\%$). In contrast, volumetric soil moisture at less than 40 cm depth correspondently increased with rainfall, although never exceeded approximately 30%. This was due to textural contrast between horizons and the subsequent capacity of sandy clay to retain more moisture than sandy loam. S14 comprised sandy loam to the sample depth of 80 cm, thus the definitive horizons recognised in S9 volumetric soil moisture values are not present. At S14, volumetric soil moisture at 10 cm depth showed a rapid wetting and drying regime, governed by the amount of rainfall received (refer Figure 5.2). Volumetric soil moisture at irrigated sites (S1, S2, S4, S6, S8, S10, S12, S13) within the woodlot never fell to the low volumetric soil moisture levels experienced at S14, as effluent application maintained a higher volumetric soil moisture at all irrigated sites.

6.2.4 Soil moisture curves

Saturation for all depths ranged between 32 – 42%. Average volumetric soil moisture (Figure 5.20) shows that these values existed at most sites for most of the study period, particularly at lower depths (< 60 cm) and profiles with B horizons. Thus at a woodlot scale, it was assumed that adequate volumetric soil moisture was maintained for much of the study period, particularly at depths of greater than 60 cm. However, several sites experienced soil moisture values less than WP for long periods. The most notable
of these are sites located upslope within the woodlot and/or sites that exist on the periphery of the woodlot, where these sites may receive less irrigation and are exposed to stronger winds that increase PET. These sites included S1 and S2 (refer Figures 5.8 and 5.9 respectively). In contrast, S5 and S10 experienced saturated conditions at depths of less than 40 cm, exceeding FC on numerous occasions throughout the study period.

6.3 Temporal soil ESP

The frequency of soil sampling allowed for the interpretation of ΔESP at a resolution not previously attempted. The data presented (Figure 5.30 and Table 5.6) showed average soil ESP at irrigated sites can fluctuate markedly within a two-month period. For example, April 2003 ΔESP values (Figure 5.30) decreased by approximately 2 %, 7 %, 10 %, 6 % and 5 % at the 10, 20, 40, 60 and 80 cm depths respectively. One of the largest accumulation periods, April - June 2002 resulted in soil ESP increasing by approximately 2 %, 5 %, 9 %, 8 % and 5 % at the 10, 20, 40, 60 and 80 cm depths respectively.

Sample sites outside the woodlot, S9 and S14, displayed much larger fluctuations over time. For example, ΔESP at S14 for the August – October 2002 period showed that soil ESP decreased by approximately 5 %, 10 %, 18 %, 17 % and 20 % at the 10, 20, 40, 60 and 80 cm depths respectively. ΔESP at S14 for the April – June 2002 period showed that soil ESP increased by approximately 11 %, 15 %, 23 %, 20 % and 24 % at the 10, 20, 40, 60 and 80 cm depths respectively. S9 also showed larger fluctuations in ΔESP over time than most study sites. The fact that volumetric soil moisture nears FC for much of the time at S9 (non-irrigated) at depths greater than 60 cm indicates that the sub-surface volumetric soil moisture values obtained at irrigated sites were not significantly different to pre-irrigation conditions.

Sites S9 and S14 have never received irrigated effluent and were subject to rainfall only. When rainfall occurred at these sites, sodium was mobilised and transported downwards through the soil profile. In the prolonged absence of rainfall, sodium became concentrated within the soil matrix as described in Chapter 2.
in rainfall on salt storage and leaching in the soil profile have been demonstrated by Smith et al. (1996).

In this study, the soil sodium levels at both S9 and S14 responded in a similar manner to the research by Smith et al. (1997), that is, they appear to be governed by volumetric soil moisture status. Smith et al. (1997) showed that sodium is readily leached from the soil after rainfall and accumulates after irrigation with effluent. The non-irrigated sites in this study experienced more severe dry periods over time because they did not receive irrigation, thus leached sodium at a greater rate after rainfall. Sodium became more concentrated at S9 and S14 than irrigated sites because an artificially increased hydraulic load was not leaching them. As a result, soil ESP at S9 and S14 exhibited larger fluctuations over time, which was directly attributed to the timing and depth of rainfall.

Figure 5.30 and Table 5.6 show irrigated sites experienced less fluctuation, as they became chemically accustomed to frequent effluent application. However, it was assumed that the Outside Solution (OS) within pore space between clay particles contained significantly higher sodium concentrations. Over time, the result was that the residence time of the effluent/rainfall in the root zone at irrigated sites was greater than that of the OS at non-irrigated sites.

Coupled with relatively higher volumetric soil moisture values, sodium was continually moved downward through the profile of irrigated sites by advection as demonstrated in the column-leaching experiments. This allowed sufficient time for Na⁺-enriched soil water and micro-aggregates in the root zone to come closer to chemical equilibrium. Thus, the range of ΔESP values for irrigated sites were buffered against the larger ΔESP range shown at non-irrigated sites.

The non-irrigated site S9 was compared with the irrigated site S10 for interpreting sites with duplex horizons. While the fluctuation of soil ESP between sampling periods is generally greater at the non-irrigated site, actual soil ESP remained generally lower than that of the irrigated site. However, soil ESP for the non-irrigated site showed similar maxima at various times throughout the study period.
The fact that the effluent irrigated site had relatively higher soil ESP meant the primary source of sodium was from secondary treated effluent applied to the woodlot. The source of sodium for non-irrigated sites would be weathering and dissolution of the soil over time and the amount contributed by rainfall. The actual sodium loading applied to the effluent irrigated woodlot (Area 3) will be discussed in Section 6.3.2.

Most research on soil sodium in effluent irrigation projects has sampled soil on an annual basis, or longer, as typified in studies such as Balk et al. (1998) and Hulugalle and Finlay (2003). Other studies investigated the effect of sodium on soil properties using vast numbers of varying soil types throughout a given region (McIntyre, 1979; Powell et al., 1995; Nelson et al., 2002; Vance et al., 2002). These results provide valuable inventory on various soil types, although little on the changing response of these soils to changes in ESP and effluent/rainfall SAR over time.

The implication for managing sodium loading is highlighted in the dynamic nature of sodium flux in an effluent irrigated woodlot. This study has shown that soil ESP varies by as much as 24% over a four-month period and therefore the timing of soil sampling and analyses are relevant in managing soil ESP. Soil varies in clay content, where for a given clay content only a finite number of exchange sites exist in a given volume of soil. Consequently, soil ESP should be determined with respect to the sodium flux by increasing the frequency of soil sampling after a pre-determined sodium load has been applied, in order to assess leaching rates.

Change in sodium (cmol(+)/kg) provided the necessary insight into actual sodium variation within the soil profile at different sites. The cation flux of Ca$^{2+}$, Mg$^{2+}$, Na$^{+}$, K$^{+}$ and CEC at various depths from February 2002 to October 2003 (Figures 5.30 to 5.35) did not always reflect ΔESP values, particularly during periods of cumulative irrigation surplus/deficit deficit. During periods of cumulative irrigation surplus/deficit surplus (upward sloping trends in Figure 5.4), the cation flux of Ca$^{2+}$, Mg$^{2+}$, Na$^{+}$, K$^{+}$ and CEC at various depths from February 2002 to October 2003 better reflected ΔESP values. This was because soil water movement was nearer to steady-state flow, as indicated by generally higher volumetric soil moisture values.
6.3.1 Site comparisons: irrigated sites versus non-irrigated sites

Monitored trends were compared between sites of similar texture and profile. As a result, S1 (irrigated) and S14 (non-irrigated) were chosen because each had loamy sand to a depth of 80 cm, whilst S10 (irrigated) and S9 (non-irrigated) were selected because each had a loamy sand A-horizon and a sandy clay B-horizon. The comparison of volumetric soil moisture between sites of similar texture and profile are shown in Figures 1 and 2.

S1 was the irrigated site compared to the non-irrigated site S14. The obvious expectation was that S1 would have higher volumetric soil moisture due to effluent irrigation. This was the case and was particularly evident in the uppermost 10 cm of S1. S14 had very low volumetric soil moisture at depths of 10 cm, compared to S1. However, volumetric soil moisture at lower depths at S1 were not only less than those at S14, but were less than, or near to, wilting point for much of the study period. Soil ESP was generally higher for S1 than S14, at all soil depths. Both sites show an increase in soil ESP with depth, which was similar to other sites monitored during this study. The higher soil ESP at S1 was attributed to effluent irrigation and the excess cations it contains. However, during June-August 2002, soil ESP maxima at S14 are similar to soil ESP at S1, for all depths. The significance was that the “non-irrigated” site variation can be comparable to the irrigated site, despite the non-irrigated site not receiving effluent irrigation.

S14 exhibited lower soil ESP values than S1 for the majority of the study period, although S14 displayed a greater variation in ΔESP, with leaching regimes driven by rainfall only. In contrast, S1 ΔESP values fluctuated less because additional cations in the effluent and the soil micro-aggregates attained a chemical equilibrium similar to that of the applied effluent. Large rainfall events (June and December 2002) leached cations through all depths at both S1 and S14, indicating that soil structure was not altered to significantly decrease permeability at these times.
Figure 6.1: Summary of soil ESP and $\theta$ for S1, S8 and S14 (soils with loamy sand A and B horizons)
Figure 6.2: Summary of soil ESP and $\theta$ for S5, S10 and S9 (soils with sandy clay B horizons)
S. A. Lucas

S10 was the irrigated site compared to the non-irrigated site S9. A higher $\theta$ was expected at S10 because of effluent irrigation and this was generally the case for the upper 40 cm of the profile. In contrast, volumetric soil moisture values at the non-irrigated site S9 exceeded those of S10 at depths of less than 40 cm, despite not receiving irrigation. The reason for this is likely due to the absence of trees that could extract water at these depths.

At S10, volumetric soil moisture values at 40 cm depth were greater than volumetric soil moisture values at 60 cm depth for the majority of the study period. This may indicate the presence of an effluent irrigation-induced, perched water table. Evidence suggests that as soil water was impeded by the sandy clay B horizon at 20 cm, additional effluent has resulted in “marsh” conditions. Over the past 12 years, bullrushes, sedges and other reeds have established themselves in the saturated conditions, highlighting the change caused by effluent irrigation over time. Prior to effluent irrigation, these species were not common to the planned woodlot, due to the sandy nature and high hydraulic conductivities of Area 3 soils (HWC, 1991).

Volumetric soil moisture at S9 showed a distinct difference between upper and lower depths because of the A and B horizons present. Whilst effluent irrigation at S10 has masked the presence of these horizons, volumetric soil moisture at 40 cm depth remains higher than the corresponding depth at S9. Furthermore, volumetric soil moisture values at S10 never approached zero while volumetric soil moisture values at S9 did near zero on several occasions. S9 was only “re-wetted” by rainfall whereas S10 received both effluent irrigation and rainfall.

Soil ESP at S9 was generally lower than soil ESP at S10, however, on many occasions the soil ESP at both sites were comparable, particularly under relatively higher volumetric soil moisture. The $\Delta$ESP for the non-irrigated site S9 was found to be greater than $\Delta$ESP at S10. $\Delta$ESP at S9 was greater than $\Delta$ESP at S10 for two likely reasons:

i) Volumetric soil moisture at S9 showed greater variability than volumetric soil moisture at S10, thus a greater relative sodium flux would be expected from prolonged wetting and drying cycles.
ii) The soil structure at S10 has deteriorated to a stage of minimal permeability with respect to the soil ESP/effluent SAR relationship. This was indicated by the relative stability of volumetric soil moisture values over time and the presence of perched water tables. With respect to micro-aggregate/soil pore stability, further discussion is made with reference to S10, the $C_{TU}$ and the $C_{TH}$ in Section 6.4.4.

The comparison of cation ratios shown in Chapter 5.3.5 showed significant variability between the loamy sand and sandy clay horizons for Ca/Mg and Na/Mg values. The relationship between soil ESP and Ca/Na was a strong one (as expected) however the variability in the others was not expected. The variation in the loamy sand horizon was likely due to the variation in chemistry of applied effluent over time and the low clay content and CEC of this horizon. The presence of excess cations due to irrigation, compared to non-irrigated sites, provide the source of variability. The lack of clay particles prevented exchange of cations and many could be considered entrained cations existing between soil pore spaces in the A horizon.

Conversely, very little variation occurred in the sandy clay horizons, which was likely due to the soil matrix (predominantly clay with lower relative permeability than loamy sand) and relatively higher CEC of this horizon. In addition, the clay particles in the sandy clay (horizon B) could be assumed to be saturated with cations originating from effluent after 13 years of application. As a result, the soil profile was assumed to be in equilibrium with the volume-weighted average SAR of applied waters and saturated with respect to the cations present at the time. Therefore, the sandy clay horizon of the identified clay mineralogy (montmorillinite) will preferentially displace and exchange cations as shown in Kopittke et al. (2005).

Kopittke et al. (2005) used solutions of SAR ranging from 1 – 80 mmol(+)/L and four ionic strengths between 10 – 200 mM. For montmorillinite, the study showed that preference for $Na^+$ increases as soil ESP increases under solutions of small ionic strengths (10 – 50 mM) and shows a preference for $Ca^{2+}$ at higher ionic strengths (200 mM). This means that when the B Horizon (sandy clays) soils reached a threshold soil
ESP there was likely to be a preferential exchange in favour of Ca$^{2+}$, resulting in displaced sodium being transported further downward through the soil profile.

The threshold soil ESP for preferential displacement of Na$^+$ in favour of Ca$^{2+}$ was likely to occur at soil ESP values greater than 20 as shown in Figure 5.7 (ESP v Ca/Na ratio). At lower soil ESP values, increases in ESP are likely to be buffered by increased Ca$^{2+}$ as shown in Figure 5.7 (ESP v Ca/Mg ratio). At higher soil ESP values, further increases in ESP are likely to be buffered by increased Mg$^{2+}$ as shown in Figure 5.7 (ESP v Na/Mg ratio). The sodium flux observed during the study period may have been partially driven by these processes, however closer investigation of these phenomena was not undertaken and warrants further research with respect to selecting appropriate sites for effluent irrigation schemes.

**6.3.2 Sodium loading**

The two largest sodium loading periods were in May/June 2002 and December 2002/January 2003. Each period had at least five irrigation events that supplied greater than 40 kg Na+/ha to the woodlot soil. Subsequent soil sampling after these periods resulted in the largest increases in soil ESP, at all depths, for January 2002 – October 2003. Other periods where irrigation was frequent, yet rarely exceeded 20 mm, maintained a sodium-leaching regime at high $\theta$ values. This leaching regime was seen whereby some depths experienced loss/gain in soil ESP, then a gain/loss the following soil-sampling period. This was particularly apparent for soil ESP over depth in June, August and October 2003.

The large range of sodium loads applied through irrigation and the soil-sampling program (two-monthly) made it difficult to interpolate loss/gain in soil ESP on a daily time-step basis. For STE irrigated sites, the soil ESP and $\Delta$ESP show significant fluctuation over time. Additional sodium was sourced from STE irrigation, as the non-irrigated sites receiving only rainfall showed lower soil ESP, particularly for sandy loam horizons.
Lateral movement of sodium in the soil profile was assumed to be at a minimum under low $\theta$ and at a maximum under high $\theta$, with the greatest potential for off-site mobilisation occurring when the ground water level at GW2 was less than 0.4 m from the surface. However, overall lateral sodium movement was assumed negligible for the majority of the study period, as no surface ponding on the lower adjacent property was observed. Periods where lateral movement could have potentially occurred included those where cumulative irrigation surplus/deficit was in surplus (above the line in Figure 5.5).

Soil sodium and $\theta$ are highly dynamic in the soil profile over time and with depth and the time-steps used for some monitoring parameters did not adequately capture the dynamic nature of the system, particularly with respect to $\theta$. The weekly $\theta$ values obtained were insufficient to interpolate daily changes in $\theta$ with respect to daily STE irrigation scheduling or rainfall events. Despite this, the trends observed in cumulative irrigation surplus/deficit are validated by the WATSKED model soil water predictions ($r^2 = 0.92$, refer Figure 5.1 in Chapter 5). Therefore, temporal variations in the monitored parameters will be discussed to provide better insight into various conditions that could deleteriously impact upon soil structure in an effluent irrigated woodlot soil.

6.3.3 Interpretation of temporal variations in monitored parameters

This section discusses the interaction of major trends with respect to applied STE and rainfall (Figure 5.2), the cumulative irrigation surplus/deficit (Figure 5.5), volume-weighted SAR (Figure 5.21), average volumetric soil moisture (Figure 5.20), soil ESP and $\Delta$ESP (Figure 5.30), groundwater depth (Figure 5.6) and sodium loading (Figure 5.38). The change in sodium (cmol(+)/kg) graph is not reproduced, although CEC and cation flux of $\text{Ca}^{2+}$, $\text{Mg}^{2+}$, $\text{Na}^+$, $\text{K}^+$ were discussed in Section 6.3.2.

The study period (January 2002 – October 2003) was divided into sub-periods, which have been shown in Figure 6.3. The sub-periods were created using the major increasing and decreasing trends observed in the cumulative irrigation surplus/deficit. Figures (5.21, 5.30) [see Figure 6.5] and Figures (5.2, 5.6 and 5.38) [see Figure 6.6] (respectively) have also been reproduced on pages 177 - 178 for ease of comparison.
during discussion. Volumetric soil moisture for irrigated/non-irrigated sites with Loamy Sand (A + B horizons) and Loamy Sand A horizons/Sandy Clay B horizons were shown in Figures 6.1 and 6.2.

Each sub-period in Figure 6.3 represents an observed trend in the cumulative irrigation surplus/deficit. In this section, averaged volumetric soil moisture and soil ESP results from sites within the woodlot are discussed in relation to the observed slope of cumulative irrigation surplus/deficit. Specific site comparisons, including non-irrigated sites were discussed previously in this Chapter.
Figure 6.3: Cumulative irrigation surplus/deficit between January 2002 and October 2003 highlighting wetting and drying trends.
Figure 6.5: The volume-weighted average SAR of applied STE and rainfall (June 2002 and October 2003) and the Average soil ESP and ΔESP for all sites within the woodlot (S1, S5, S8, S10) (February 2002 – October 2003)
Figure 6.6: Applied effluent; Rainfall; Depth to groundwater monitored on a weekly basis (May 2002 – November 2003); and Monthly sodium loading (June 2002 – October 2003)
The soil ESP/effluent SAR relationship is referred to in terms of the 0 – 10 cm depth only, as the impact of low SAR rainfall is the most damaging to soil structure on the surface. By the time rainwater permeates below 10 cm, the SAR has increased and was less “aggressive” in compromising micro-aggregate/soil pore stability. The column-leaching by depth experiment in Section 5.3.9 showed this and is further discussed in Section 6.4.1.

**Sub-period 1: 8/2/02 – 3/4/02**

In Figure 6.3, the 8/2/02 – 3/4/02 sub-period showed decreasing cumulative irrigation surplus/deficit that ranged from 80 mm to approximately -100 mm. The average volumetric soil moisture for this sub-period ranged between 18 % to 30 % at 10 cm, 24 % to 34 % at 40 cm, 28 % to 37 % at 60 cm and 32 % to 39 % at 80 cm (Figure 6.4). Average volumetric soil moisture at all depths increased during this sub-period due to excessive rainfall in February 2002, with all depths within the soil nearing field capacity by mid to late March 2002. The volume-weighted average SAR of the applied waters and groundwater depth were not available for this sub-period of the study.

When average volumetric soil moisture was greater than or equal to soil field capacity, soil ESP was expected to decrease by leaching and, according to Figure 6.5, this appeared to be the case. The April 2002 ΔESP values show that soil sodium remained similar to February 2002 in the upper 10 cm, increased by approximately 1 % at 20 cm, whilst all other depths recorded some decrease (range -1.5 % to -4 % in Figure 5.30). Optimum permeability would have been decreased due to rainfall SAR being less than C_{TU} and as a result, downward movement of solutes through the soil profile also decreased.

When the area was wet, maximum soil volumetric soil moisture and minor surface ponding were observed in Area 3. This occurred during February 2002 and effluent irrigation ceased throughout March 2002. Since zero effluent or rainfall occurred at the site during March 2002, the calculated cumulative irrigation surplus/deficit naturally decreased. However, PET does not cease and resulted in a month of irrigation deficit when calculated from equation 11. This in turn results in a
downward sloping cumulative irrigation surplus/deficit (soil deficit trend), although volumetric soil moisture values gave the true soil water condition at each site.

**Sub-period 2: 3/4/02 – 29/6/02**

In Figure 6.3, the 3/4/02 – 29/6/02 sub-period showed increasing cumulative irrigation surplus/deficit (wetting period) that ranged from approximately -100 mm to approximately 200 mm. Figure 6.3 shows that the average volumetric soil moisture for this sub-period ranged between 24 % to 32 % at 10 cm, 31 % to 34 % at 40 cm, 32 % to 38 % at 60 cm and 32 % to 40 % at 80 cm. Volumetric soil moisture at all depths remained ≥ FC for some sites during this sub-period and surface ponding was observed in the region of S5, S10 and S8 (refer Figure 3.4 for location).

Effluent irrigation was increased substantially during this sub-period, particularly during April and May 2002 (Figure 6.6). A significant rainfall event in late May 2002 increased cumulative irrigation surplus/deficit to approximately 200 mm, the highest surplus recorded for the entire study period. During this time, groundwater levels increased to their maximum heights at all bores (Figure 6.6).

Average soil ESP for April – June 2002 increased at all depths (Figure 6.5). Rainfall (SAR = 0.5) would have altered micro-aggregate/soil pore stability at 0 – 10 cm, as the CTU was 2.3 (soil ESP = 13), although the decrease in permeability was not observed in the field. Several rain events occurred during June 2002, which were “piggy-backed” by a week of effluent irrigation.

In addition, whenever sufficient soil water was present, highly soluble sodium would have been hydrodynamically dispersed through moist interstitial pore space and in the direction of soil water movement (Shaw, 1994). Also, processes such as bioturbation and preferential soil water flow around tree roots enhance any advective solute movement through the soil profile (White, 1997), while it is commonly known that forest soils exhibit greater permeabilities than cropped soils (Hewlett, 1982; Dingman, 1994). As a result, sodium was transported downward through the saturated soil profile, even under lower permeability.
Sub-period 3: 29/6/02 – 8/12/02

In Figure 6.3, the 29/6/02 – 8/12/02 sub-period showed decreasing cumulative irrigation surplus/deficit (drying period) that ranged from approximately 200 mm to approximately -480 mm. Figure 6.4 showed that average volumetric soil moisture for this sub-period ranged between 12 % to 29 % at 10 cm, 23 % to 31 % at 40 cm, 30 % to 36 % at 60 cm and 33 % to 40 % at 80 cm.

The frequency of irrigation was increased to promote sodium leaching and maintain optimum permeability. As a result of increased irrigation and minimal rainfall between August and November 2002, the volume-weighted average SAR of the applied waters averaged 5.4 (Figure 6.5). This shows optimum permeability, as an SAR of 5.4 lies between the C\textsuperscript{TU} and the C\textsuperscript{TH}.

From August – November 2002, groundwater levels decreased which was also reflected in the decreasing average volumetric soil moisture at all depths. During this time, evidence suggests that PET exceeded hydraulic loading (daily average PET for the study period = 4.6 mm). The average daily hydraulic loading for this sub-period was 1.8 mm and zero surface ponding or runoff was observed.

From August – November 2002, it would be expected that soil ESP would be increasing due to potential evapotranspiration and evaporative concentration. Figure 6.5 suggested this, as a positive $\Delta$ESP occurred for two consecutive sampling periods (April 2002 - June 2002 and June 2002 - August 2002). The $\Delta$ESP shows that sodium increased at all depths and, summed from both soil-sampling intervals, ranged from 7 % to 12 %.

During October/November 2002, additional irrigation was applied to encourage leaching of the accumulated sodium experienced during this sub-period. In addition, several rainfall events in early December 2002 increased average volumetric soil moisture at all depths in the woodlot, showing rapid permeability at all depths. These rainfall events were complimented with effluent irrigation, which increased the volume-weighted average SAR. This allowed for optimum permeability to be maintained and as result, zero surface ponding was observed, even after the 80 mm rainfall event in early December 2002.
Sub-period 4: 8/12/02 – 17/1/03

Increasing cumulative irrigation surplus/deficit (Figure 6.3) occurred as a result of increased irrigation and rainfall up until 8/12/02 (refer Figure 6.6). In Figure 6.3, the 8/12/02 – 17/1/03 sub-period showed a fluctuating cumulative irrigation surplus/deficit, that moved from approximately -380 mm, through -410 mm, then increased to -390 mm. Figure 6.4 showed that the average volumetric soil moisture for this sub-period ranged between 15 % to 28 % at 10 cm, 23 % to 32 % at 40 cm, 27 % to 38 % at 60 cm and 37 % to 39 % at 80 cm. All soil depths recorded a decrease in average volumetric soil moisture at the start of this sub-period, particularly in the uppermost 40 cm, then average volumetric soil moisture at all depths declined throughout sub-period 4.

The volume-weighted average SAR of the applied waters averaged 4.6 (Figure 6.5), therefore preventing dispersion (\( \gg C_{TU} \)). Permeability at this time was assumed to be optimum since the average volumetric soil moisture for all depths increased within approximately three weeks of the rising cumulative irrigation surplus/deficit, showing relatively rapid permeability to the sampled depth of 80 cm. Soil ESP was expected to decrease under these conditions. Figure 6.5 suggested this was the case, as a negative \( \Delta ESP \) occurred for the August 2002 to October 2002, ranging from -3.5 % to -8 %. During October 2002 to December 2002, results show sodium accumulation at less than 40 cm and leaching at lower depths, due to average volumetric soil moisture decreasing at depths of less than 40 cm.

Groundwater depth remained relatively stable at all bores during this sub-period. Even with the relatively larger volumes of effluent applied during sub-period 4, minimal variation in groundwater depth occurred. This suggested that even when the irrigation system was operating at a maximum delivery rate, and the soil at optimum permeability, groundwater levels were not drastically affected by effluent irrigation. The only time groundwater levels were significantly increased was after intense and/or prolonged rainfall and this was indicative of larger catchment hydrology effects rather than influence from effluent irrigation.
Sub-period 5: 17/1/03 – 17/2/03

In Figure 6.3, the 17/1/03 – 17/2/03 sub-period showed decreasing cumulative irrigation surplus/deficit (drying period) that ranged from approximately -390 mm to approximately -490 mm. Figure 6.4 showed that the average volumetric soil moisture for this sub-period ranged between 14 % to 16 % at 10 cm, 22 % to 25 % at 40 cm, 27 % to 28 % at 60 cm and 33 % to 34 % at 80 cm. As a result of low cumulative irrigation surplus/deficit and low average volumetric soil moisture, the soil ESP would be assumed to be increasing. The short duration of this sub-period made it difficult for interpretation as no soil sampling occurred.

Sub-period 6: 17/2/03 – 5/6/03

In Figure 6.3, the 17/2/03 – 5/6/03 sub-period showed increasing a cumulative irrigation surplus/deficit (wetting period) that ranged from approximately -490 mm to approximately -260 mm. Figure 6.4 showed that the average volumetric soil moisture for this sub-period ranged between 18 % to 28 % at 10 cm, 25 % to 28 % at 40 cm, 28 % to 32 % at 60 cm and 32 % to 36 % at 80 cm. All depths experienced increasing average volumetric soil moisture due to constant effluent irrigation and intermittent rainfall events through this sub-period.

Soil sampling occurred in the last week of the month and as a result, ΔESP actually reflected the conditions of sub-period 5. Figure 6.5 shows ΔESP values ranged from 3 % to 8 % for the December 2002 - February 2003 soil sampling interval. Irrigation was increased between late February 2003 and May 2003 (see Figure 6.6), which was showed by the increasing cumulative irrigation surplus/deficit in Figure 6.3 and the slowly rising average volumetric soil moisture values in Figure 6.4. Significant rainfall occurred during this time, although irrigation continued at a lower rate in an attempt to maintain a relatively higher volume-weighted average SAR to promote optimum permeability.

The ΔESP ranged from -3 to -14 % for the February 2003 - April 2003 sampling interval (Figure 6.5) and represented the largest decrease for any two-month period and subsequently for the entire monitoring period. From April 2003 until June 2003 the cumulative irrigation surplus/deficit was increasing (Figure 6.3). This provided optimum
conditions, that is, relatively high average volumetric soil moisture, increasing cumulative irrigation surplus/deficit and suitable volume-weighted average SAR of application waters, for promoting solute movement through the woodlot root zone.

For the same interval, the volumetric soil moisture in the upper 10 cm fluctuated significantly while average volumetric soil moisture at lower depths appeared to have remained relatively stable (Figure 6.4). The volume-weighted average SAR of the application waters was approximately 3 during May/June 2003, whilst the 10 cm depth had a soil ESP of 11.5 % (see Figure 6.5). Since the $C_{TU}$ would be approximately equal to 3, then a decrease from optimum soil permeability would be expected.

Groundwater levels generally increased throughout sub-period 6, although it was difficult to determine the contribution to increases in groundwater levels between effluent irrigation and catchment hydrology effects. Situated on alluvial deposits, the woodlot studied is approximately 1.32 hectares and was estimated to represent a negligible percentage of the total catchment influenced by rainfall. Therefore, any significant rainfall had a greater probability of influencing groundwater levels than any effluent applied to the woodlot. Furthermore, groundwater EC varies dramatically in the Hunter Valley and is dependent on the geology of the groundwater source (Beale et al., 2000).

Sub-period 7: 5/6/03 – 1/10/03

In Figure 6.3, the 5/6/03 – 1/10/03 sub-period showed decreasing cumulative irrigation surplus/deficit (drying period) that ranged from approximately -390 mm to approximately -600 mm. Figure 6.4 showed that the average volumetric soil moisture for this sub-period ranged between 12 % to 26 % at 10 cm, 23 % to 28 % at 40 cm, 28 % to 32 % at 60 cm and 34 % to 37 % at 80 cm. Although generally decreasing, average volumetric soil moisture exhibited an erratic trend from July to late September 2003, particularly at 10 cm depth. During this time, the irrigation system experienced problems (algal blockages) and coupled with several intermediate rainfall events resulted in minimal and sporadic STE application. The erratic nature of average volumetric soil moisture at 10 cm depth may reflect this (Figure 6.4). Actual trends in volumetric moisture for individual sites are shown in Figures 6.3 and 6.4.
The interesting aspect of Figure 6.5 was that after increased irrigation, \( \Delta \text{ESP} \) values displayed a cyclic loss/gain for the last three soil sampling intervals (April 2003 – June 2003, June 2003 – August 2003, August 2003 – October 2003) (Figure 5.30). \( \Delta \text{ESP} \) data within these soil sample periods exhibit positive values for some depths and negative values for others. Positive \( \Delta \text{ESP} \) for April 2003 – June 2003 (10, 20, 40 cm) became negative \( \Delta \text{ESP} \) for June 2003 – August 2003, whilst the negative \( \Delta \text{ESP} \) for April 2003 – June 2003 (60, 80 cm) became positive \( \Delta \text{ESP} \) for June 2003 – August 2003.

This showed sodium movement through the root zone, particularly from the less than 40cm to the greater than 40 cm depths, and probably reflects poor matching of daily irrigation scheduling with daily PET. Also, this could be a direct result of light rainfall events not permeating to lower depths, then being “piggy-backed” during successive irrigation episodes of slightly larger magnitude. If so, then this appears to show the sodium accumulation or leaching “front” moving through the root zone, the rate of which was governed by the slope of plotted cumulative irrigation surplus/deficit over time and validated by average volumetric soil moisture values.

Summary

In this study, observed relationships between cumulative irrigation surplus/deficit, the sodium flux (\( \Delta \text{ESP} \)), volume-weighted average SAR of effluent/rainfall, volumetric soil moisture (\( \theta \)), groundwater levels and the soil ESP/effluent SAR continuum for micro-aggregate/soil pore stability, were as follows:

i) An increasing cumulative irrigation surplus/deficit and increasing average volumetric soil moisture promoted sodium leaching – dominantly by advection and under high soil ESP values (> 20). Under these conditions, groundwater levels provide the limiting factor for additional hydraulic loading.

ii) A decreasing cumulative irrigation surplus/deficit and decreasing average volumetric soil moisture suggested sodium accumulation – dominantly by evaporative concentration. Under these conditions, the volume-weighted average
SAR of the application waters and the ESP of the receiving soil governed the response of soil structure in maintaining optimum permeability.

iii) An increasing cumulative irrigation surplus/deficit and decreasing volumetric soil moisture indicates sodium accumulation, although the accumulation “front” will be relative to the depth of effluent applied (Figure 2.10, Section 2.3.3). Under these conditions, PET was the limiting factor for hydraulic loading.

iv) A decreasing cumulative irrigation surplus/deficit and increasing average volumetric soil moisture represents poor permeability. Under these conditions, the soil profile was saturated and did not allow additional hydraulic loading to permeate below the root zone. Indicative of effluent SAR being less than the \( C_{TU} \) of the receiving soil and/or a clay horizon present near to the surface.

v) The perpendicular distance of the cumulative irrigation surplus/deficit from the soil saturation line (Figure 6.3) reflects lost irrigation opportunities (deficit) or surplus irrigation in excess of the trees’ requirements. For example, in January 2003 the cumulative irrigation surplus/deficit was – 400 mm (Figure 6.3). This showed that the opportunity to irrigate 400 mm of effluent has been missed.

vi) Excessive sodium loading (frequent irrigation events of > 20 mm/day for two consecutive months) caused higher soil ESP for longer periods, despite increasing volumetric soil moisture trends. This occurred as soil water “equilibrated” to the chemistry of the irrigation waters.

All of the above points should be considered when managing an effluent irrigated woodlot. The ability to leach sodium and maintain suitable soil water in the woodlot root zone will depend on management strategies that encourage optimum permeability by maintaining soil structure. The soil ESP/effluent SAR continuum for micro-aggregate stability provides the framework for these decisions and is discussed in Section 6.4.2.

With respect to the soil ESP/effluent SAR relationship, results have shown that the effluent irrigated woodlot soil was susceptible to dispersive conditions on numerous occasions from January 2002 – October 2003. Trends in sodium accumulation and leaching indicate that adequate micro-aggregate/soil pore stability was maintained at
many of the monitored sites within the woodlot. This was particularly evident when large rainfall events (SAR < C\textsubscript{TU}) leached sodium through to depths of 80 cm.

6.4 Implications of sodium flux for managing effluent irrigated woodlots

This section discusses the implications for irrigated woodlot soil management with respect to the sodium flux. The sodium flux has been shown to be greater than previously recognised and in terms of the relationship between soil ESP and effluent SAR, indicates that some aspects of current site assessment and monitoring may miss the potential benefits for adaptive soil management.

6.4.1 Column-leaching experiments

Aim 3 of this study was to examine the implications for STE irrigation scheduling and soil management when using waters of varying SAR to a woodlot soil of varying soil ESP. Results from column-leaching experiments showed the response of the woodlot soil with respect to dispersion and flocculation that could not be observed in the field. However, whilst care was taken to pack soil columns of equivalent bulk density, the use of packed soil columns was likely to enhance the observed response, namely:

- The displacement/exchange of cations from the soil exchange surface resulting from waters of varying SAR and subsequent variations in micro-aggregate/soil pore stability;
- The translocation of clay particles downward through the soil profile; and
- The relative variations in soil permeability.

The main purpose of these experiments was to observe relative changes in soil structure, specifically in relation to the SAR of the application waters and the subsequent turbidity of the output solution from the columns. However, if accurate hydraulic conductivity measurements of in-situ soils are to be determined, then soil cores sampled in-situ should be used. However, these experiments were conducted to show the difference in permeability rates due to the effects of increasing Na\textsuperscript{+}. Therefore, whilst the comparison of hydraulic conductivity is not a true reflection of in-situ soil hydraulic
conductivity, the experiment did encourage saturation of all soil pores and observation of reduction in pore size due to the variation in cationic charge driven by variation in permeant chemistry (volume-weighted average SAR); and the potential for dispersion and subsequent translocation of clay particles and/or blocking of deformed soil pores (micro-aggregate/soil pore stability). This phenomenon governs the deviation from optimum permeability for a given soil chemistry and effluent irrigation scheduling condition.

**Column-leaching in relation to the \( C_{TU} \) and \( C_{TH} \)**

In Figure 5.42, Column 1 experienced optimum permeability as the SAR of the effluent applied was between the \( C_{TU} \) and the \( C_{TH} \). Micro-aggregate/soil pore stability was not compromised in this column, thus a clear permeant was observed. Column 2 experienced poor permeability because the SAR of the applied rainwater was less than \( C_{TU} \) and caused dispersion. Micro-aggregate/soil pore stability was altered and permeability decreased due to dispersion. This resulted in dispersed clay particles appearing in the permeant of Column 2 and was consistent with the change in permeability results predicted by the work of Quirk and Schofield (1955).

**Column-leaching by depth**

Turbidity shows the presence of dispersed clay, where the downward movement of dispersed clay into soil macropores has been identified by many researchers as the dominant process decreasing permeability at ESP less than 25 (Cass and Sumner, 1982; Rengasamy *et al.*, 1984; Emerson, 1991; So and Aylmore, 1993; Halliwell *et al.*, 2001; Quirk, 2001). The six parameters monitored during these experiments and presented in Section 5.4.1 will now be discussed.

The pH and EC of the output solution during the 0 - 10 cm series showed an inverse correlation (\( r^2 = 0.63 \), Appendix 9), meaning the change in pH and EC of both the effluent and the intermittently applied rainfall were strongly related in the way they alter the potential cation exchange/displacement conditions within the soil profile. pH and EC for the 10 - 20 cm series showed a weaker negative correlation (\( r^2 = 0.45 \), Appendix 10). In the 0 – 10 cm series, the relative charge of the applied rainwater
increased in favour of $H^+$ (decreasing pH), cation mobilisation was enhanced and an increase in leached cations was expected, as shown in the work of Chorom and Rengasamy (1995). As the relative charge of the applied effluent decreased in favour of alkali cations (increasing pH), cation mobilisation was decreased and a decrease in leached cations was expected.

For a STE irrigated woodlot soil, the change in relative charge conditions of the soil/soil water matrix helped explain the lower fluctuations in $\Delta$ESP at irrigated sites when compared to non-irrigated sites. For example, rainfed non-irrigated sites are never artificially enhanced with respect to cations, thus exhibit larger fluctuations in $\Delta$ESP due to continual wetting/drying by rainfall of low cation concentration (low EC) and subsequent leaching. Conversely, STE irrigated sites buffer $\Delta$ESP due to the continual presence of excess cations introduced to the soil profile through irrigation as initial soil ESP at STE irrigated sites was higher.

The EC and SAR showed a strong correlation for the 0 - 10 cm series ($r^2 = 0.94$) and a poor correlation for 10 - 20 cm series ($r^2 = 0.16$). The change in correlation reflected the change in cation composition between being eluted through the 0 – 10 cm series, then being introduced to the 10 – 20 cm series. The 10 – 20 cm series displayed SAR values much higher than initial application waters, which was attributed to the leaching of exchangeable cations (seen in the change in cation exchange capacity (CEC)) from the 0 – 10 cm series (Table 5.10). In addition, the 10 - 20 cm series appeared to have retained approximately 80% of the cations leached from the 0 - 10 cm series. At this stage, it must be noted that input samples to the 10 - 20 cm series may have contained the dispersed clay particles from the 0 - 10 cm series and these could potentially block macropores at the surface of the 10 - 20 cm series column.

Changes in permeability ranged from approximately 55 to 140 mm/hr for the 0 – 10 cm series and 35 to 90 mm/hr for the 10 - 20 cm series (Figure 5.43D). For the 0 - 10 cm series, the initial soil ESP was 21.6 (Table 5.10, Section 5.4.1), which gives a $C_{TH}$ and $C_{TU}$ of 12.7 and 3.7 respectively. The initial SAR of the rainwater was 0.5; therefore dispersed clay translocated down the profile and decreased permeability. Upon addition
of the STE at SAR 5.2, micro-aggregates became more stable (less dispersion), although permeability still decreased due to osmotic swelling (swelling by excess Na\(^+\)).

Figure 5.43D showed a decrease in permeability over time, although the trend constantly fluctuated throughout the experiment. The erratic nature was believed to be due to the constant head/falling head conditions of the experiment. When the volumetric flask was inverted and suspended above the column, the first 150 mL was applied under constant head conditions. When the flask emptied, approximately 100 mL permeated under falling head conditions. However, there was a distinctive decreasing trend in permeability over time, which was consistent with results from past studies (Quirk and Schofield, 1955; Agassi et al., 1981; Rengasamy and Olsson, 1993; So and Aylmore, 1993; Crescimanno et al., 1995; Magesan et al., 1999).

Figure 5.43E showed the turbidity of the output solutions over time. Upon addition of rainwater, there was an increase in turbidity (SAR < C\(_{\text{TU}}\)). However, over time the amount of dispersed clay decreased with each subsequent rainwater addition. This was attributed to additional cations increasing SAR within the column, present from previous STE output samples from the 0 – 10 cm series. Addition of STE at SAR 5.2 (SAR > C\(_{\text{TU}}\)) did improve stability, as turbidity decreased to less than 1 NTU, although permeability remained lower due to the presence of excess sodium in the column.

Both results are consistent with the pioneering work of Quirk and Schofield (1955). As a result, a relatively stronger negative correlation existed between turbidity and SAR (\(r^2 = 0.46\), see Appendix 9) for the 0 - 10 cm series output samples. Of particular interest was that a similar correlation existed for the 10 - 20 cm series output samples (\(r^2 = 0.51\), see Appendix 10). The turbidity and SAR relationship was the only correlation consistent over time and depth for the column-leaching experiments.

The decrease in CEC indicates that clay particles translocated downward into Column 2 also transport bound cations through the 0 – 10 cm depth. This was significant because the CEC at subsequent depths will appear to alter (increase in cations). The number of cation exchange sites will also increase if clay percentage increases, as was the case with the 10 – 20 cm columns.
Solute-breakthrough curves were established based on the work of several scientists (van Genuchten and Alves, 1982; Shackelford, 1994, 1995). The assumptions made were consistent with those discussed during Section 2.2.6. The solute-breakthrough curves (Figure 5.43F) show that sodium was readily transported through the soil profile, particularly in the 0 - 10 cm series. Figure 5.43F shows a series of breakthrough curves on the one axis, where there are three curves for the 0 – 10 cm series. The exact time of breakthrough is difficult to obtain, however, the solute curve plateaus in a relative short time after the STE addition.

The 10 - 20 cm series was difficult to interpret, as the sodium concentration varied as a result of the 0 - 10 cm series output samples, which were of mixed initial concentrations. Peaks do occur in the 10 - 20 cm series curve and the slopes can be interpreted as a secondary solute break-through curve. This curve also showed rapid sodium transport through the 10 - 20 cm series, although not as rapid as observed in the 0 – 10 cm series.

Results from column-leaching experiments show a thorough understanding is needed of the cation exchange characteristics of a soil proposed as an STE disposal site before a meaningful definition can be made for optimum permeability. Optimum permeability is the range of SAR values that exist between the $C_{TU}$ and the $C_{TH}$ of a given soil at a given time. Results show how dynamic this relationship is with respect to sustainable effluent irrigation. Therefore, the concept of optimum permeability is useful for managers of effluent irrigation projects when appropriate monitoring and management strategies are implemented.

Using disturbed soil columns introduces an uncertainty in accurately determining field permeability conditions. Common field methods for determining permeability in-situ involve using solution-filled instruments, like a disc permeameter. This experiment showed that any permeability tests performed in the field must incorporate the effect of solution SAR and soil ESP. For example, a permeability test should use a range of SAR solutions when assessing the suitability of a soil of known ESP, for use with irrigation waters of known SAR. Therefore, the significance of the $C_{TU}$ and $C_{TH}$ should be used in unison when assessing a woodlot soil of specific chemical and textural properties for receiving STE irrigation of specific SAR.
The universal response of soil of specific ESP being irrigated by waters of specific SAR was validated by Quirk (2001, p1214), in that:

“Experience with Riverina clay and Cajon sandy loam indicates that the basic physico-principles with respect to the threshold and turbidity concentrations are readily applied in the field”

With respect to the behaviour of montmorillinite, the dominant clay mineral found in the studied soils, it was assumed that at high soil ESP values and low electrolyte concentrations the clay particles would preferentially select Na$^+$ over Ca$^{2+}$ with a decrease in that preference as soil ESP decreases and electrolyte concentration increases, as shown in the work of Kopittke et al. (2005). With the abundance of Mg$^{2+}$ available for exchange in the studied soils, it is likely that preferential exchange of Mg$^{2+}$ at low soil ESP and high electrolyte concentrations accounted for decreases in soil ESP, particularly in the lower sandy clay horizons.

In summary, the main results emanating from these experiments show:

i) that excess sodium was not tightly bound to the exchange complex and was transported readily;

ii) that changes in SAR of the permeating waters impact upon micro-aggregate stability, causing clay particles to move from the 0 – 10 cm depth to the 10 – 20 cm depth, contributing to decreased permeability at both depths;

iii) that the micro-aggregate/soil pore stability of the woodlot soil exposed to STE was “re-stabilised”, although with lower permeability;

iv) that SAR and turbidity were the only parameters to have a strong relationship over depth;

v) the soils’ response to sodium loads was highly dynamic and extremely difficult to capture in the field.

To interpret these results, an understanding of the soil ESP/effluent SAR continuum for micro-aggregate/soil pore stability is required.
6.4.2 The soil ESP/effluent SAR continuum and the Emerson Aggregate Test.

The Emerson Aggregate Test (EAT) (Emerson, 1991) is a common test using distilled water as an immersion solution in determining soil structural stability. Soil consultants commonly use the Emerson Aggregate Test (Emerson, 1991) to assess the suitability of soil structure for potential effluent irrigation sites. Fundamentally, the EAT (Emerson, 1991) is a test that imparts energy to promote instability, and relate the observed stability of micro-aggregates within the soil profile.

In comparing this continuum to the EAT (Emerson, 1991), micro-aggregates with an ESP much greater than $C_{TH}$ represent class 7 aggregate stability and micro-aggregates with an ESP less than $C_{TU}$ represent class 1 aggregate stability. Other positions between the $C_{TH}$ and $C_{TU}$ give the classes and sub-classes between 1 and 7 (Loveday and Pyle, 1973; Emerson, 1991), although no boundaries between each class have been defined. As a result, the continuum can be used in conjunction with the EAT (Emerson, 1991) using immersion solutions over a range of differing SAR values to predict soil response to application waters.

The key concepts for performing this test involve using small aggregates of known size and mass. However, whether the aggregate has similar ESP to another aggregate of similar size and mass is impossible to determine (non-destructively) before placement into the immersion solution. Therefore, the interpretation of aggregate stability should ultimately be determined in light of the SAR of the immersion solution used during the test.

The EAT (Emerson, 1991) for a given soil can give different results depending on the SAR of the immersion solution. With respect to STE irrigated woodlots, this is significant in that some soils that have an Emerson Class 2 aggregate stability in distilled water may in fact become more stable under a low strength STE with an SAR greater than the $C_{TU}$ of the receiving soil. This implied stability also has implications for the amelioration of soils rich in exchangeable sodium, although these are outside the scope of this study. Relevant research on this topic has been detailed in the work of Qadir et al. (2000).
6.4.3 Hypothetical application of soil ESP/effluent SAR Continuum

In this hypothetical application, a wastewater manager wants to dispose of secondary treated effluent to a nearby field containing pasture grasses. The receiving soil has an ESP value of 10 and a corresponding SAR value of 10 (ESP = SAR for values < 32) and the low strength STE to be applied has an SAR value of 6. Permeability tests have been performed and the receiving soil was suitable for the proposed effluent irrigation scheme.

Using Equations 5 and 6, the $C_{TH}$ and $C_{TU}$ can be calculated as 6.2 and 1.8 mmol/L respectively. The continuum indicates that at an SAR value of 6, the applied effluent does indeed fall between the “existing soil structure” and the “$C_{TH}$”, therefore implying that the effluent was electrolytically suitable for optimum permeability.

Problems occur in the later years, after sodium has accumulated in the soil profile and/or irrigation source waters change in chemical composition. Assuming that source waters remained stable in their chemical composition, the ESP value had risen to 35 over a 6-year period. Since the receiving soil was sampled annually (hypothetically), there has been no way of knowing whether the soil ESP value of 35 was representative of the range of temporal soil ESP that existed for any given irrigation season.

The lack of data acquired over a six-year period does not greatly assist in describing temporal variations in soil sodium flux or how the accumulation could have been managed. Monitoring rates of environmental change at a resolution great enough to describe the various flux of components within the system, was paramount in defining trends and ranges of the monitored parameters over time. Quite often, the effluent SAR will remain constant, or increase relative to the increase in salts of the source waters (Patterson, 1994).

If the effluent SAR value remains at 6 and the soil ESP value has increased to 35, then the recalculated $C_{TH}$ and $C_{TU}$ will be 20.2 mmol/L and 5.8 mmol/L respectively. Since the SAR value of the effluent was approximately equal to $C_{TU}$, the continuum was driven towards dispersion, resulting in surface ponding and/or runoff, as the applied effluent may not permeate. When rainfall occurs, the SAR falls to $<< C_{TU}$ and dispersion will occur. Prior to these times, appropriate adaptive management can minimise
“dispersive” events. For example, introducing Ca$^{2+}$ to the “system” would decrease soil ESP over time and maintain soil structure.

### 6.4.4 Application of the continuum to this study

Figure 6.3 through Figure 6.8 show soil ESP for the 0 – 10 cm depth as a function of the $C_{TU}$ and $C_{TH}$ for irrigated sites within the Branxton woodlot (S1, S5, S8, S10) versus the volume-weighted average SAR of the application waters from January 2002 – October 2003. Dots represent the volume weighted SAR of the application waters for that day. The lower solid line represents the $C_{TU}$ and the upper dotted line represents the $C_{TH}$. Volume-weighted average SAR values lying between the $C_{TU}$ and $C_{TH}$ represent optimum permeability.

Volume-weighted average SAR values above the $C_{TH}$ indicate a decrease in permeability (> 15 % decrease) (Quirk and Schofield, 1955), although micro-aggregate/soil pore stability was being maintained. Volume-weighted average SAR values lying below the $C_{TU}$ indicate that dispersion was taking place to some extent and degrading micro-aggregate/soil pore stability, the extent of maximum instability was represented by Column 2 results in Figure 5.42. Permeability may decrease to zero under these conditions resulting in surface runoff and/or prolonged surface ponding.

Figure 6.7 shows the number of hydraulic events at S1 where the SAR was below the $C_{TU}$, between the $C_{TU}$ and $C_{TH}$ and above the $C_{TH}$. Approximately 50 % of all events had SAR values less than $C_{TU}$. From a management point of view, it is desirable to irrigate during these events to maintain optimum permeability by increasing the volume-weighted average SAR of the permeating waters. This cannot always be achieved. For example, a rain event could be of such magnitude as to saturate the site. At the Branxton woodlot this occurred in early February 2002 and June 2002 (refer Figure 5.2). Similar volumes of rainfall occurred in early to mid December 2002, although the site did not become saturated.

At this time the cumulative irrigation surplus/deficit was very low (~ -450 mm, Figure 6.1) and the soil water storage was assumed to have had the capacity to receive the rainfall volume at that time. Further irrigation during January 2003 failed to increase volumetric soil moisture at 40 cm and 60 cm, although irrigation did increase volumetric
soil moisture at 10 cm and 80 cm (refer Figure 5.9). The relatively lower volumetric soil moisture at 40 cm and 60 cm was attributed to PET from the root zone over the summer period.

Figure 6.7: Branxton woodlot (S1 - Area 3) – $C_{\text{TU}}$ and $C_{\text{TH}}$ versus the volume-weighted average SAR of application waters Jun02 – Oct03.

Figure 6.8: Branxton woodlot (S5 - Area 3) – $C_{\text{TU}}$ and $C_{\text{TH}}$ versus the volume-weighted average SAR of application waters Jun02 – Oct03.
Figure 6.9: Branxton woodlot (S8 - Area 3) – $C_{TU}$ and $C_{TH}$ versus the volume-weighted average SAR of application waters Jun02 – Oct03.

Figure 6.10: Branxton woodlot (S10 - Area 3) – $C_{TU}$ and $C_{TH}$ versus the volume-weighted average SAR of application waters Jun02 – Oct03.
Figure 6.7 shows approximately 50% of all events had SAR values > $C_{TU}$ but ≤ $C_{TH}$. This suggests optimum permeability during these events. Due to the sandy texture of the site soils, permeability was assumed to be relatively high. Although permeability was not directly measured in this study, a previous geo-technical report assessed Area 3 soil as having adequate soil water storage capacity (~80 – 160 mm) and high hydraulic conductivity (HWC, 1991).

Figure 6.8 shows the number of hydraulic events at S5 where the SAR was below the $C_{TU}$, between the $C_{TU}$ and $C_{TH}$ and above the $C_{TH}$. Approximately 35% of all events had SAR values less than $C_{TU}$. Figure 6.8 shows approximately 65% of all events had SAR values > $C_{TU}$ but ≤ $C_{TH}$ and no events exceeded $C_{TH}$.

Figure 6.9 shows the number of hydraulic events at S8 where the SAR was below the $C_{TU}$, between the $C_{TU}$ and $C_{TH}$ and above the $C_{TH}$. Approximately 25% of all events had SAR values less than $C_{TU}$. Figure 6.9 shows that approximately 50% of all events had SAR values > $C_{TU}$ but ≤ $C_{TH}$ and approximately 25% of all hydraulic events exceeded $C_{TH}$. At S8, volumetric soil moisture at depths of ≤ 40 cm remained high from February 2002 to October 2003 (refer Figure 5.14) and was the first site to experience waterlogged conditions after excessive rainfall and/or irrigation. In addition, S8 was topographically the lowest point in the woodlot and sub-surface soil water would theoretically “drain” through this point.

Figure 6.10 shows the number of hydraulic events at S10 where the SAR was below the $C_{TU}$, between the $C_{TU}$ and $C_{TH}$ and above the $C_{TH}$. Approximately 50% of all events had SAR values less than $C_{TU}$. Figure 6.10 shows approximately 50% of all events had SAR values > $C_{TU}$ but ≤ $C_{TH}$ while zero events occurred that exceeded $C_{TH}$.

In summary, S1 and S10 appear to have the greatest potential for dispersion, as many of the hydraulic events have volume-weighted average SAR values near to or less than $C_{TU}$. Management options should include soil amelioration strategies that would increase $Ca^{2+}$ at these sites in an attempt to re-establish soil structure by decreasing soil ESP. This should encourage sodium leaching and replenishment of $Ca^{2+}$ to the soil matrix, at least in the woodlot root zone. The effect would be a result similar to S8.
(Figure 6.9), where a reduced SAR, caused by increased Ca\(^{2+}\), would stabilise micro-aggregate/soil pore structure with respect to the C\(_{TU}\) and the C\(_{TH}\).

Decreased permeability, prolonged surface ponding and well-defined sodium leaching regimes were expected and this was not necessarily observed from results in the field. However, the potential for dispersive conditions and the possible response of woodlot soils may take the direction of site S10, where it does appear that soil structure has altered over time resulting in saturated conditions and relatively high soil ESP values.

In addition to specific sites requiring more attention than others, field results from this study show that soil ESP can increase and decrease significantly over a four-month period. Increases in soil ESP for a two-month period were comparable to other studies at other sites that observed similar increases over years (Smith et al., 1996; Falkiner and Smith, 1997; Myers et al., 1999; Meneer et al., 2001). Results from this study emphasised the dynamic nature of sodium in the soil profile and the increased potential for soil structural problems if it is not properly managed. When applied to the practice of irrigation of secondary treated effluent to land, this imposes severe constraints on the sustainability of effluent land disposal.

Such constraints include suitable soil sodium management, which at this point in time, appears to be based on an estimated leaching fraction and periodic soil assessment. This in part is due to the priority outcome of the irrigation scheme, for example, increasing crop yield, decreasing water use or preventing groundwater contamination. No irrigation scheme appears to specifically address sodium management as a priority, unless the irrigation scheme is an amelioration strategy for sodium removal.

Irrigation schemes are not specifically designed to promote sodium leaching or maintain soil structure. To emphasise the potential for soil structure failure, Quirk (2001) refers to the work of McGeorge and Fuller (1950), which predicted one application of the research by Quirk and Schofield (1955). This application has not been fully appreciated in the literature on effluent irrigation schemes. During a shortage of Colorado River water between 1946 – 48, irrigation water was pumped from an underground supply (McGeorge and Fuller, 1950; Quirk, 2001). Cited in Quirk (2001, p1212):
“This water contained 50 mmol(+)/L of sodium and 8 mmol(+)L of calcium plus magnesium (SAR = 17.7). As a result of the use of this water for 3 years the average exchangeable sodium percentage was increased to 25. When the river water with an electrolyte concentration of 3.9 mmol(+)/L became available for laboratory use and was used the soil “froze up”, from which it may be reasonably showed that the concentration of electrolyte in the river water was less than the turbidity concentration since its use caused such a drastic alteration of the soil structure.”

Although the Colorado River water had an electrolyte concentration slightly higher than the calculated $C_{\text{TU}}$ (2.7 mmol(+)/L), Quirk states that an exact correspondence was not expected using another soil and an estimated SAR value (Quirk, 2001). Nevertheless, this soil response could be expected to occur in all irrigated areas that alternate between dam, river, effluent or groundwater sources.

Bond (1998) iterates that there are very few truly sustainable practices in the land application of secondary treated effluent, but does encourage further research to increase the lifetime of effluent irrigated sites by working more closely with wastewater managers, consultants and soil scientists. To do this, the variation in the response of soils of varying ESP to irrigation waters of varying volume weighted average SAR values must be simplified. The soil ESP/effluent SAR continuum for micro-aggregate/soil pore stability presented in this study contributes to the simplification of anticipating the response of soil structure, particularly with respect to dispersion and flocculation within the soil profile.

With respect to sustainability, the “lifetime” of the site will depend on the volume and SAR of the effluent applied and of course, different wastewaters will be more suitable to some soils than others. It is envisaged that using the soil ESP/effluent SAR continuum as a STE irrigation management tool, coupled with routine, statistically sound soil sampling techniques (Rayment, 1985) and analysis (Rayment & Higginson, 1992), will improve the long-term physical and chemical structure of any soil receiving STE.
6.4.5 Recommended strategy for managing sodium flux

The observed sodium flux was used to develop an adaptive management strategy at Branxton based the daily and cumulative irrigation surplus/deficit, volumetric soil moisture, soil ESP, effluent SAR and variations in groundwater depth. The initial and boundary conditions for each were set using their monitored data over time. The sodium flux has shown greater variability than past studies at other sites would show. This variability has ramifications for soil structure over time, thus the sustainability of STE irrigation to woodlots. Objectives of the management strategy are:

- To provide a simple adaptive irrigation strategy that promotes optimum soil permeability conditions in order to better manage increasing sodium in woodlot soils receiving STE over time.
- To incorporate the water balance, soil exchange characteristics and the soil ESP/effluent SAR relationship, as all can impact on relative soil permeability.

Exclusive use of irrigation scheduling has been shown to be ineffective in managing sodium flux and maintaining soil structure over the long term. Figure 6.9 depicts a management flow diagram that assesses on a daily basis, any potential dispersion event based on accessible monitoring data such as ground water levels and soil moisture. Soil ESP would most likely be based on 3-monthly intervals, with that value used for the following three month period.

A soil dispersion event is caused when the applied waters have an SAR less than $C_{TU}$, which impacts upon micro-aggregate/soil pore stability and reduces permeability. A series of temporal dispersion events may occur over a number of years, leaving the soil unsuitable for future alternate land use, particularly when soil ESP increases and/or source irrigation waters change in chemical composition.

For sustainable effluent irrigation to woodlot soils, it is desirable to minimise or alleviate dispersive conditions. For example, the addition of $Ca^{2+}$ to the system reduces the impacts of excess $Na^+$ in the soil profile although increases the total electrolyte concentration of the system. This maintains soil structure, thus productive soils, and allows woodlot irrigation management to concentrate on the water balance component (primarily) whilst acting to prevent excessive sodium accumulation.
Figure 6.11 shows a flow diagram for irrigation scheduling to minimise the frequency of dispersive events, which is described by a series of YES/NO options, and is explained as follows:

1. The cumulative irrigation surplus/deficit determines irrigation scheduling. If the answer is YES at this stage, continue to stage 2. If the answer is NO, then go to stage 3. \( \Delta I \) can be determined from Equation 11.

2. Calculate the volume of effluent/rainfall required to satisfy PET and go to stage 4. If there is an irrigation surplus, any additional hydraulic load should promote deep drainage if optimum conditions apply (go to stage 4).

3. Assess the need to adjust the cumulative irrigation surplus/deficit and check volumetric soil moisture. If irrigation is required (low volumetric soil moisture, decreasing cumulative irrigation surplus/deficit) go to stage 2. If not (high volumetric soil moisture, increasing cumulative irrigation surplus/deficit) go to stage 8.

4. Soil ESP is converted to an SAR and used to determine \( C_{TU} \) and \( C_{TH} \). If the volume-weighted average SAR of effluent + rainfall (Qe + Qp) is less than the \( C_{TU} \), go to stage 6. If (Qe + Qp) is > \( C_{TU} \), then go to stage 5.

5. Groundwater depth may be of consequence if near to the surface, particularly with respect to phosphorus and nitrogen from effluent irrigation. In addition, the depth to groundwater may indicate the potential for off-site salinity issues. If groundwater depths are deemed suitable, go to stage 7.

6. Soil ESP and/or effluent SAR can be amended to be electrolytically suitable and promote optimum permeability. This predominantly involves increasing Ca\(^{2+}\) in the system. If SAR amendment can be achieved, go back to stage 5. If this cannot be undertaken, go to stage 8.
Figure 6.11: Flow diagram for irrigation scheduling to minimise the frequency of soil dispersion events.

1. Is daily ΔI in deficit?
   - Y: Calculate site (Qe + Qp) to equal daily PET, then ask:
   - N: Y

2. Is the volume-weighted average SAR of (Qe + Qp) < C_TU?
   - Y: Can this be amended to be > C_TU?
   - N: Is groundwater at a critical depth to prevent irrigation?

3. Is irrigation required to adjust ΔCI or increase θ?
   - Y: Irrigate accordingly
   - N: Y

4. Is daily ΔI in deficit?
   - Y: Y
   - N: X

5. Is groundwater at a critical depth to prevent irrigation?
   - Y: Irrigate accordingly
   - N: Y

6. Can this be amended to be > C_TU?
   - Y: Irrigate accordingly
   - N: X

7. Commence irrigation. Having satisfied the PET with respect to the daily and cumulative irrigation surplus/deficit and soil structure, the optimum permeability should be better maintained over time with this approach with respect to increasing sodium applied through effluent irrigation.

8. Do not irrigate. If irrigation is applied under the given conditions, either the hydraulic load will be greatly overestimated and/or soil structure will suffer prolonged “dispersive” conditions, particularly in the upper 10 cm. In any case, sustainability will be substantially compromised if not properly managed. Large storm events result in the hydraulic load being naturally exceeded and due to a severe reduction in SAR, rainfall will enhance dispersion and also destabilise soil structure through raindrop impact (White, 1997).
If irrigation is deemed unviable for soil structure, the implementation of effluent amendment strategies and/or soil management strategies should be investigated.

This management flow diagram allows wastewater managers to implement adaptive strategies for irrigation scheduling that will promote sodium leaching and maintain PET. This is a result of maintaining optimum soil permeability with respect to soil ESP and effluent SAR. In application, micro-aggregate/soil pore stability is qualified using the continuum for micro-aggregate/soil pore stability (Figure 5.44), although the interpretation of common tests used to determine this stability, such as the EAT, are more complex than they first appear when applied to effluent irrigated woodlot soils.
Chapter 7: CONCLUSIONS AND FUTURE RESEARCH

This study aimed to examine temporal sodium dynamics (flux) in a woodlot soil receiving secondary treated effluent in order to investigate the effect variations in soil ESP and effluent/rainfall SAR have on long-term relative soil structure, in particular, the ramifications for soil management. Temporal trends in water balance data and soil chemistry were determined and, coupled with column-leaching experiments, were used to develop an effluent irrigation strategy to maintain optimum permeability with respect to sodium. The implications for soil management and STE irrigation scheduling have also been discussed.

Weekly volumetric soil moisture was monitored over the longer term with the purpose of predicting the capacity of the cumulative irrigation surplus/deficit to receive irrigation or rainfall. When cumulative irrigation surplus/deficit was compared to the WATSKED model (Myers et al., 1999), comparable trends were obtained and it was therefore considered adequate for future water balance calculations ($r^2 = 0.92$). The effectiveness of this approach in monitoring soil sodium flux will be different for receiving soils of varying clay content.

Soils with clay contents of approximately 50% are the most difficult to manage (Shaw et al., 1994). However, the soil structure of sites in this study comprising loamy sand to the sampled depth of 80 cm were not observed to be significantly altered as to decrease permeability. Therefore, using trends in water balance components to monitor soil sodium flux requires careful interpretation.

Irrigation waters of suitable volume-weighted average SAR values (> $C_{TU}$) tend to provide optimum permeability for the receiving soil and reduce soil ESP through leaching if relatively high $\theta$ can be maintained. Irrigation waters of unsuitable volume-weighted average SAR (< $C_{TU}$) tend to reduce permeability of the receiving soil and increase soil ESP. The temporal variation in volume-weighted average SAR of the application waters ranged from 0.5 (indicative of rainfall only) to approximately 5.9 (indicative of effluent only) and using the continuum for micro-aggregate/soil pore stability presented in this study, provided evidence that some hydraulic application
events were more susceptible to causing dispersive conditions than others, especially during frequent and intense rainfall to soils of relatively higher soil ESP.

The principal outcome was that the temporal sodium flux at the study site was greater than could have been previously recognised. Proven theories were then used to model a soil ESP/effluent SAR continuum for micro-aggregate/soil pore stability in the soil profile. This in turn can be used as an adaptive management tool for assessing optimum permeability for soils receiving secondary treated effluent. The implication is that the potentially dispersive irrigation/rainfall events leading to decreased permeability can be predicted, assessed and managed before further loss of soil structure occurs.

At the Branxton woodlot (Area 3), the maximum temporal soil sodium flux occurred over a four-month period and varied between sampling intervals by as much as 24 % (February 2003 – May 2003). The cyclic loss and gain of ESP (sodium flux) between most sample periods indicated that the woodlot soil regularly experienced optimum permeability conditions and/or adequate irrigation scheduling; meaning the leaching of sodium downward through the root zone was maintained. Soils with higher clay contents experience greater cation exchange, thus greater variation in temporal soil ESP and the $C_{TU}$ and $C_{TH}$.

Soil ESP versus $C_{TU}$ and $C_{TH}$ show that, on average, approximately 75 % of all STE irrigation and/or rainfall had volume-weighted SAR values suitable for optimum permeability. Soil ESP versus $C_{TU}$ and $C_{TH}$ indicate several hydraulic loading events at the Branxton woodlot (Area 3) would have caused dispersive conditions, as shown by column-leaching experiments, thus a decrease in soil permeability was expected under the described water balance conditions. These simple plots are useful for decisions in soil management and timing of amelioration strategies.

Sodium flux can directly alter soil structure over time and if the range of temporal sodium flux is too great, potentially dispersive conditions are more likely to occur that may reduce permeability. The concept of sustainable effluent irrigation and soil management relies on the ability of soil to transmit applied waters and leach accumulated sodium. To achieve this, it is desirable to have an understanding of how sodium can affect optimum permeability. In conjunction with water balance monitoring, the sodium flux in a woodlot soil irrigated with STE has been shown to be greater than
previous studies would indicate. The response of the soil, with respect to permeability, to waters of varying SAR is predicted from the continuum presented.

The continuum relies on soil ESP and effluent SAR to form this prediction, thus monitoring approaches should incorporate this relationship into irrigation scheduling and soil management. Sustainable STE irrigation scheduling and soil management are closely related. The balance of cations in the receiving soil will ultimately attain that of the irrigated STE if not properly monitored. However, the introduction of Ca\(^{2+}\) (gypsum or lime) into the soil water system will improve micro-aggregate/soil pore stability by maintaining a balance of Na\(^+\) and Ca\(^{2+}\).

The variation in magnesium (Mg\(^{2+}\)) from February 2002 to October 2003 was of such magnitude on occasions as to significantly alter soil ESP, thus a true indication of sodium flux may not have been obtained. Results from cation (Ca\(^{2+}\), Mg\(^{2+}\), Na\(^+\) and K\(^+\)) and CEC flux results showed that this was the case at some sites, particularly at low \(\theta\) and where sandy clay was present. Therefore, caution should be shown when interpreting soil ESP over time, especially in a soil that has other cations present in relative excess.

Having identified the sodium flux and determined the soil ESP/effluent SAR conditions for optimum permeability, there is a need to discuss strategies for maintaining optimum permeability in an effluent irrigated woodlot. Ultimately, in conjunction with irrigation scheduling, it is the ability to maintain optimum permeability that will guarantee sodium movement through the root zone whilst reducing the probability of surface runoff due to rainfall and/or over-irrigation.

The micro-aggregate/soil pore stability of a woodlot soil receiving secondary treated effluent can be predicted using the continuum, while suitable management strategies can be implemented. These strategies would involve altering soil ESP, effluent SAR or both. The principle is to decrease the soil ESP or effluent SAR by addition of Ca\(^{2+}\). For example, farmers for over a hundred years have utilised gypsum (CaSO\(_4\)) or lime (CaCO\(_3\)) to “break-down” clay and increase permeability.
The breakdown of clay refers to flocculated clay structure rather than one that is dispersed. While flocculated clays provide a greater permeability, dispersed clay particles can cause surface sealing and decrease soil permeability to zero (Levey, 1984; Meneer et al., 2001; Rousseva et al., 2002). The presence of calcium in the soil, at least equal to that of sodium and magnesium, is considered favourable for productive agricultural soils (McBride, 1994).

According to both formulae for soil ESP and effluent SAR (equations 1 and 2, Section 2.2.3), the addition of Ca$^{2+}$ appears to be the most effective management tool available for maintaining optimum permeability. Ca$^{2+}$ was the denominator in those equations therefore an increase in Ca$^{2+}$ will cause a decrease in either soil ESP or effluent SAR. Adding gypsum to the soil is a simple approach that can be implemented by managers of the effluent reuse scheme and is particularly useful in reducing soil ESP. Under-irrigation of a woodlot with secondary treated effluent for extended periods has been shown to promote sodium accumulation, therefore excess Ca$^{2+}$ will buffer sodium flux that could have destabilised soil structure.

Optimum permeability can be maintained by increasing the SAR of the application waters, but is this the best option? The higher the soil ESP rises from effluent irrigation, the more susceptible to dispersion the soil is from rainfall ($<$ C_TU) if soil sodium is not properly managed. Therefore, gypsum (CaSO$_4$) or limestone (CaCO$_3$) “chips” of approximately 20 mm diameter could be strewn throughout the woodlot. This would have a slow-release effect and as rainfall occurred, Ca$^{2+}$ would be partially dissolved and be moved into the root zone. Not only would these “chips” act as a source of Ca$^{2+}$ to the soil, but they would also reduce soil ESP over the long-term. This was evident from soil ESP results for the irrigated site S8, which had been applied with lime in 1994. During this study, S8 still had appreciable amounts of Ca$^{2+}$ throughout the soil profile even though lime application occurred approximately 10 years ago.

To improve management of STE irrigated soils with respect to soil ESP/effluent SAR, it is envisaged that the main monitoring parameters and approach used in this study be developed to acquire daily time-step or continuous data. For example, instruments that can continuously monitor groundwater depth, volumetric soil moisture, soil EC and the water balance do exist and could be used to better assess individual
hydraulic loading events. From the monitored water balance components, only cumulative irrigation surplus/deficit and hydraulic load were monitored on a daily time-step, while continuous monitoring of other parameters was outside the budget for this study.

It is conceded that the results presented have a number of limitations due to the monitoring frequency of some parameters, the inherent spatial variability in texture, soil moisture and chemistry of the soil itself, and the lack of statistical relationships in the non steady-state system studied. Nevertheless, the resolution of the acquired data provides some limited insight into temporal sodium flux and the direction soil management must take.

The results of this study give greater insight into the potential response of soils targeted to receive irrigation, particularly if rapid changes in source waters/soil ESP are anticipated. The cumulative irrigation surplus/deficit used in this study showed a strong correlation with WATSKED model trends ($r^2 = 0.92$), although recommended STE irrigation volumes were over-estimated in light of $\theta$ at the time of irrigation. Potential reasons for this included the overestimation of the crop factor ($K_c$) used in this study and the wrong assumption of effluent distribution to Area 3 and Area 2 (at the Branxton WWTW not monitored in this study).

The concept of WATSKED (Myers et al., 1999) used to generate cumulative irrigation surplus/deficit trends in this thesis has since been revised for saline water (Theiveynathan et al., 2004), although has yet to be tested for scheduling irrigation with saline water. Testing of the WATSKED model with saline irrigation water over time, using approaches and concepts from the research presented may be the next logical research step.

Whilst not neglecting the more intrinsic properties of complex hydrosalinity models (Simunek and Suarez, 1997), this study showed that suitable proxy-indicators, such as effluent SAR and soil ESP, can provide adequate data to assess the potential micro-aggregate stability (hence optimum permeability) of the receiving soil from any irrigation and/or rainfall event. With respect to flocculation, dispersion and optimum permeability, the soil ESP/effluent SAR continuum for micro-aggregate/soil pore
stability can be used to predict the structural response of woodlot soils irrigated with effluent and subsequently improve the efficiency of irrigation scheduling and adaptive soil management.
REFERENCES


