UNIVERSITY OF SOUTHERN QUEENSLAND
Faculty of Health, Engineering and Sciences

Effect of Dispersed Clay and Soil Pore Size on the Hydraulic Conductivity of Soils Irrigated with Saline-Sodic water

A dissertation submitted by

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I dedicate this work to my mother, father, siblings, wife and children

Awedat Musbah Awedat

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Abstract

A shortage of good quality water has led to the use of low quality, high saline-sodic water that was considered unsuitable for irrigation purposes in the past. Using such water can increase the potential of soil degradation, soil pore blockage and consequently reductions in soil saturated hydraulic conductivity (K_s). This can limit crop growth in the long term through the impacts on water and nutrient availability.

Most studies use reductions in soil K_s as an indicator of soil pore blockage when saline-sodic irrigation water is used. However, the use of saline-sodic irrigation water requires improved understanding of the of cation exchange processes, soil structural degradation and the potential for soil pore blockage. In order to address this three laboratory trials were conducted to investigate the effect of soil pore size on soil cation exchange processes, dispersed clay movement and the mechanisms of soil pore blockage in relation to changes in soil K_s.

An initial study was conducted on two soils packed into soil cores at two different bulk densities (ρ_b) and leached with solutions containing varying concentrations of clay sediments. Measurements of soil K_s showed that increasing soil ρ_b and the concentration of clay sediment in the applied solution significantly increased the percentage of clay sediment retained within the soil columns. This in turn significantly decreased soil K_s particularly for compacted soils. Soil pore blockage occurred near the soil surface in compacted soils while soil pore blockage occurred at depth in soil packed at low ρ_b. This confirmed that soil pore size has a significant effect on dispersed clay movement and potential soil pore blockage.

The soil pore size distribution does not provide information about the mechanism of soil pore blockage. It was hypothesised that clay sediment migration and soil pore blockage occurs in saline-sodic soils and that the soil response will vary depending on clay mineralogy and the quality of irrigation water applied. These issues were addressed in later experiments that used saline-sodic irrigation water applied to soil with different properties. A resin impregnation method was also used to gain a better understand the mechanism of soil pore blockage.

Solutions with different sodium adsorption ratios (SAR) were used to understand the relationship between ion exchange and changes in soil K_s for three soils. A Red Ferrosol (RF), Grey Vertosol (GV) and Black Vertosol (BV) were packed into soil cores at two ρ_b (1 and 1.2 g cm^{-3}). The rate of ion exchange and reductions in soil K_s during leaching processes were found to be significantly higher in soils with higher proportions of soil macropores (> 30 µm) compared to those dominated by micropores (< 0.02 µm). Further, the correlation between the rate of ion exchange and changes in soil K_s was stronger for the RF soil compared to the BV soil. This indicated that dispersion was the main mechanism for soil K_s reductions in the RF soil while swelling contributed significantly to reductions in soil K_s for the BV soil. Even though the reductions in soil K_s were greater in lower ρ_b soils, they maintained significantly higher K_s values compared to compacted soil. The measurement of soil exchangeable cations after leaching with high SAR solutions showed a significant increase in soil ESP at the soil surface. The RF soil also reached chemical equilibrium earlier than the GV and the BV soils.

Applications of rain water to the soil cores post-leaching with saline-sodic water led to leaching of soluble cations and significant reductions in the electrical conductivity of the soil solution and soil K_s. These significant reductions were observed just after the first pore volume (PV) of leaching. However, the RF soil maintained higher K_s

values compared to the GV and BV soil. Soils with lower ρ_b maintained higher K_s values and produced higher concentrations of dispersed clay in the leachate compared to compacted soils.

Image analyses of resin impregnated RF soil taken at depth from soil cores irrigated with good quality water (GQW) and saline-sodic irrigation water showed significant soil pore blockage after the application of saline-sodic water. This confirmed that dispersed clay migrated with infiltrated water. In compacted RF soil the soil pore blockage took place near the soil surface. For the BV soil, pore blockage occurred at and near the soil surface for ρ_b = 1 and 1.2 g cm^{-3}, respectively. Soil micropores were completely blocked while incomplete blockage was observed in the soil macropores. After applying rain water to soil cores post-leaching with saline-sodic water significant pore blockage occurred at the soil surface in all treatments except for RF packed at low ρ_b where soil pore blockage was only observed in deeper soil layers.

Soil K_s was positively correlated with increasing the soil porosity and soil macroporesity while a strong negative correlation was found between K_s and soil microporosity. Small changes in soil porosity and soil macropores resulted in significant changes in K_s. This research has shown soil pore size has a significant effect on cation exchange processes, dispersed clay movement and the potential for soil pore blockage. However, this interaction differs between soils and is dependent on the dominant clay mineralogy.

Certification of Dissertation

I certify that the ideas, designs, experimental work, software, results, analyses and conclusions presented in this dissertation are entirely my own effort, except where otherwise indicated and acknowledged.

I further certify that the work is original and has not been previously submitted for assessment in any other course or institution, except where specifically stated.

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List of Symbols and Abbreviations

BV	Black Vertosol
CEC	Cation exchange capacity
DDL	Diffuse double layer
EC	Electrical conductivity
EC_{iw}	Electrical conductivity of irrigation water
ESP	Exchangeable sodium percentage
$EXCa^{2+}$	Exchangeable calcium
EXK^{+}	Exchangeable potassium
$EXMg^{2+}$	Exchangeable magnesium
$EXNa^{+}$	Exchangeable sodium
GQW	Good quality water
GV	Grey Vertosol
K_s	Saturated hydraulic conductivity
LSD	Least significant difference
PV	Pore volume
ρ_b	Bulk density
RF	Red Ferrosol
RSC	Residual sodium carbonate
SAR	Sodium adsorption ratio
SC	Sediment concentration
TDS	Total dissolved salts
TEC	Threshold electrolyte concentration

Chapter 1: Introduction

1.1 Background

Agriculture uses approximately 65% of the total annual water consumed in Australia (Australian Water Resources Council, 2005). Yet the volume of water available for agriculture is under threat as a result of population growth, increased industrial use and recurring drought. Because of this, the focus has shifted towards the use of marginal water such as saline groundwater, drainage water and treated wastewater for the expansion of irrigated agriculture.

Poor quality water or marginal water has been successfully used for irrigated agriculture in various countries (Al-lahlam et al. 2003; Aiello et al. 2007). However, long-term impacts of such irrigation practices on soils, vegetation and surface or groundwater are variable. Salinity and sodicity are the principle water quality concerns in irrigated lands receiving this kind of water (Oster and Schroer, 1979; Ayers and Tanji, 1999) Saline-sodic water is characterised as having excess salts with sodium as the dominant cation, which increases the amount of sodium in the soil solution relative to the other major cations (Ca^{2+}, Mg^{2+} and K^{+}). These waters have high sodium adoption ratios (SAR) which is known to be associated with adverse impacts on soil aggregate stability, infiltration and saturated hydraulic conductivity (K_s). Sodium cations tend to disperse clay when they dominate the surface area of the clay, due to the monovalent nature of their charge and their relatively large hydrated radius. Halliwell et al. (2001) described the behaviour of clay particles in relation to the dominate ions present in the soil solution. The negative permanent charges on the clay particles leads to a diffuse double layer formation, in which cations cluster to neutralize the surface charge at the interface between surface clay particles and the bulk soil solution. Monovalent cations such as sodium are less attractive than divalent cations such as calcium resulting in a wider diffuse double layer (Figure 1.1). Dispersion will occur when the individual clay plates move, and remain, far enough apart to override the attractive forces as shown in Figure 1.1 (Sumner, 1993; Narasimha and Mathew, 1995; Shainberg and Levy, 2005 and Rachad and Dultz, 2007). Osmotic pressures created by the difference in ion concentrations between to the diffuse double layer and the bulk soil solution might act to expand clay lattices, causing soil swelling. In an unconfined system, this results in aggregate breakdown (Sumner, 1993).

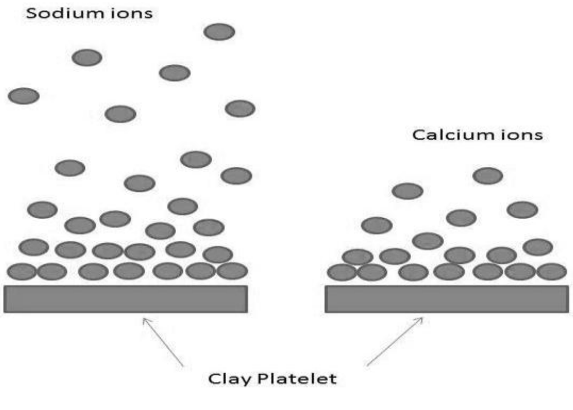


Figure 1-1 Distribution of Na+ and Ca2+ ions adsorbed on a clay surface (Hanson et al. 1999)

The effect of clay swelling and dispersion on soil hydraulic conductivity, where saline-sodic water has been applied, is widely documented (Abu-Sharar et al. 1987; Shainberg et al. 1992). Minhas and Sharma (1986) studied the effect of applying saline-sodic water with different SAR values (5, 15 and 30) and total electrolyte concentrations (15, 30, 60, and 90 mmol L^{-1}) followed by distilled water on two soils (sandy loam and clay loam). A subsequent reduction in hydraulic conductivity was found due to clay dispersion in the sandy loam soil and surface sealing in the clay loam soil.

Mace and Amrhein (2001) examined the impact on hydraulic conductivity of clay dispersion and pore plugging when using saline-sodic water with an average of SAR values (1, 3, 5, and 8) and electrolyte concentrations (0, 2.5, 5, 10, 25, 50, and 100 $\text{mmol}_c\ L^{-1}$). These treatments were followed by applications of gypsum and sulphuric acid to the soil surface and then leached with distilled water. The concentration of clay in the drainage water was used to quantify clay dispersion. Soil K_s values were reduced at all SAR values due to soil pore blockage; yet surface gypsum treatments maintained relatively higher K_s values. Soil K_s reduction was more severe for water treatments having higher SAR values of 5 and 8.

Ghiberto et al. (2007) evaluated the effect of high residual sodium carbonate (RSC > 1.25 $\text{mmol}_c\ L^{-1}$) in supplemental irrigation water on soil K_s values of a clay soil. Changes in soil K_s values were associated with increased soil dispersion causing pore blockage. Approximately 51–62% of the total clay was dispersed in irrigated plots and about 36–44% in non-irrigated plots. These studies support the theory that dispersion of finer fractions of soil and pore blockage is likely to be a major reason for the reduction in soil K_s affecting soil water movement.

Park and Conner (1980) conducted laboratory determinations of saturated hydraulic conductivity and infiltration rate on four soils varying in texture from sand to clay with five saline-sodic waters. The saturated hydraulic conductivity was not affected by water quality when saline-sodic waters were applied. However, a small addition of high quality water (low salinity water) to soil previously equilibrated with saline-sodic water caused a significant decrease in soil permeability, as a result of dispersion.

Regea et al. (1997) studied the response of low and high swelling smectites to sodicity. In low swelling clays the blockage of conducting pores due to dispersed clay was the main process responsible for the reduction in saturated hydraulic conductivity under sodic conditions. However, it was found that swelling was the main process responsible for the reduction in saturated hydraulic conductivity in the high swelling smectite clays. Furthermore, Singer (1994) studied the influences of clay mineralogy on soil dispersion and found that kaolinitic soils were less dispersive than smectitic soils, while illitic soils showed intermediate dispersivity. All of these studies considered reductions in K_s values as an indicator for soil pore blockage.

It is clear that soil pore blockage by dispersed clay is a major cause of K_s reductions in saline-sodic soil. However, work is needed to understand the processes of soil pore blockage within irrigated soil profiles with saline-sodic water application. Many studies (e.g. Flowers and Lal, 1998; Pagliai et al, 2003) show that increasing soil bulk density (ρ_b) decreases soil porosity and modify soil pore size and distribution. Increasing ρ_b decreases soil porosity and produces a higher proportion of smaller pore sizes (Gregory et al. 2006). Zhang et al. (2006) found significant reductions in K_s values in high ρ_b soils resulted from a decrease in the pore volume with equivalent pore diameter > 60 µm. These changes impact the soil water characteristics, hydraulic conductivity and infiltration rates of the soil (Zhang et al, 2005). However, no studies have been found examining the impact of soil pore size changes resulting from modifying soil ρ_b on dispersed clay migration in soil irrigated with saline-sodic irrigation water.

The movement of solute through the soil has been studied from the beginning of the use of fertilizers in agriculture (Bolt, 1979). Bolt (1979) showed how exchange and adsorption reactions could be used to predict the effects of irrigation water on both the exchange and solution phases in soil. A recent study by Jalali et al. (2007) evaluated the effect of saline-sodic irrigation water (SAR=25, EC=6 dS m^{-1}) on soil sodicity and cation leaching in calcareous soils in a soil column experiment. Upon the application of saline-sodic irrigation water, exchange occurred between sodium cations in the solution and other exchangeable cations (Ca^{2+}, Mg^{2+} and K^+) on the soil surface as high concentrations of these cations were observed in the leachate. The average ESP of the two calcareous soils during leaching increased from 9 to 21 and 28.8 to 29.7 after applying seven pore volumes of the saline sodic water to the soil cores.

Under saturated flow (e.g. during K_s measurements) soil macropores are the dominant pores and present preferential flow paths. Micropores, on the other hand, contribute to a significantly smaller proportion of total hydraulic conductivity (Hillel 2004). Consequently, a more rapid leaching of cations could be expected from the soil solution in macropores and this may cause acceleration of the ion exchange processes within preferential flow paths. Conversely, cations within soil micropores may take longer to be leached and exchanged with cations in the applied solutions. This raises a question about the effect of soil ρ_b on exchange processes between cations in applied solution and exchangeable soil cations.

It is apparent that the effect of saline-sodic water of varying qualities (SAR and electrolyte concentration) on the K_s of soil with different clay content and type is widely documented. However, there is a relative paucity of studies that have been undertaken on the effect of soil pore size on dispersed clay movement and the processes of soil pore blockage in soils irrigated with saline-sodic water. There is also a lack of work on the impact of soil pore size on cation exchange processes under saturated conditions.

1.2 Overview of research

1.2.1 Research hypotheses

The hypotheses addressed in this PhD thesis are:

- Soil pore size and solution concentration influences cation exchange in the soil when saline-sodic water is used.
- Pore blockage by dispersed clay is a major cause of reductions in soil hydraulic conductivity.
- Increasing soil microporosity increases the potential for blockage by dispersed clay.
- The potential for pore blockage by dispersed clay differs between swelling and non-swelling soil.

1.2.2 Specific objectives of research

This project aims to improve understanding of processes affecting structural changes in soil irrigated with saline-sodic irrigation water so that management strategies can be identified for sustainable use of saline-sodic water. The main objectives of this research are to:

- Develop relationships between saturated hydraulic conductivity and changes in pore size distribution when soil is exposed to water with varying amount of suspended clay.
- Evaluate the effect of soil ρ_b on cation leaching, ion exchange and steady state hydraulic conductivity of soils irrigated with saline-sodic water.
- Examine the effect of soil pore size on dispersed clay migration and the potential for soil pore blockage in swelling and non-swelling soils irrigated with saline-sodic water and after applying rain water to soils post-irrigation with saline-sodic water.

1.3 Structure of dissertation

This dissertation contains seven chapters (Figure 1.1) addressing the effects of soil pore sizes on the K_s of different soils when saline-sodic waters are used. Chapter 2 provides a general review of factors affecting soil structural stability including the impact of low quality saline-sodic water on soil structure, and consequently on K_s, as affected by clay type and irrigation system used. Further, a brief review of the effect of amendments on limiting the impact of low quality water on soil structure is included.

Materials and methods used in this research are outlined in Chapter 3. Chapter 4 reports the initial laboratory trial to evaluate the effect of porosity for two soils packed at two bulk densities on changes of soil K_s when suspensions of different sediment concentration are used. This chapter also includes preliminary measurements to evaluate soil pore blockage. The second trial (Chapter 5) investigates the effect of soil ρ_b and Na^+ concentration of the applied solution on the ion exchange rate in relation to the steady state soil K_s of different soils. The effect of soil ρ_b and the application of saline-sodic water followed by rain water on soil pore size distribution, blockage and K_s of different soils are investigated in Chapter 6. The thesis concludes (Chapter 7) with a general discussion, the conclusions and recommendations for further work.

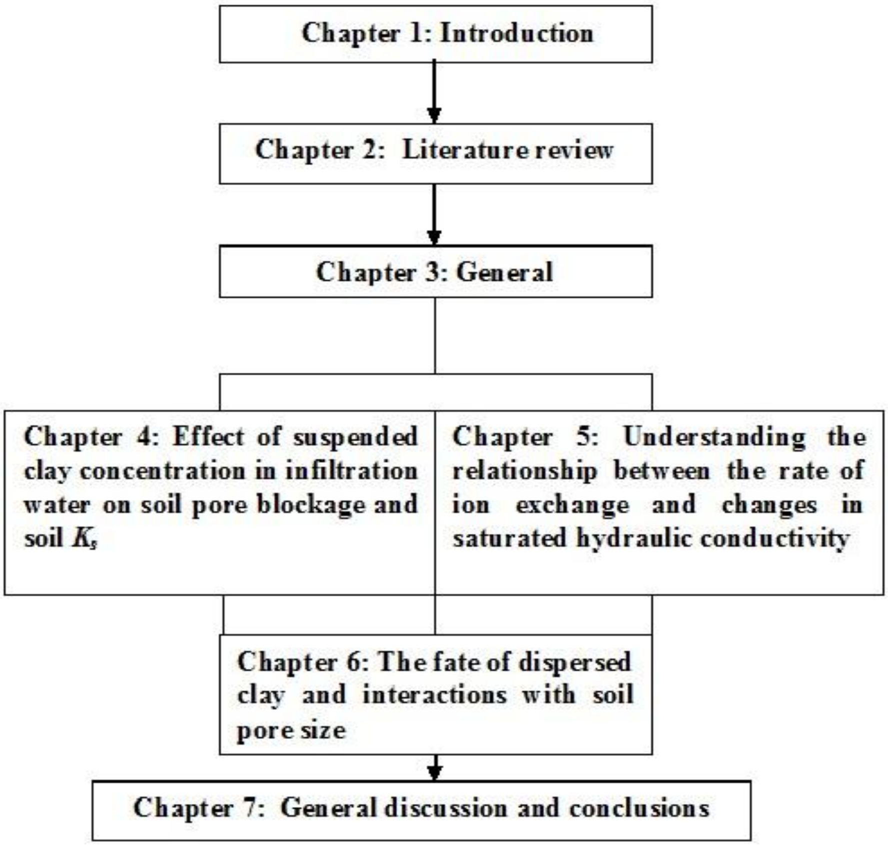


Figure 1-2 Outline of dissertation structure

Chapter: Literature Review

2.1 Introduction

The aim of this chapter is to provide a general review of factors affecting soil structure. It also gives specific consideration to problems associated with the application of saline-sodic irrigation water to irrigated soils, as well as justification for undertaking this research. This chapter consists of nine sections. Section 2.2 presents a general definition and importance of soil structure, and factors affecting soil structure. Section 2.3 discusses irrigation water salinity, sodicity assessments and their interactions with soil. Section 2.4 discusses the processes of ion exchanges within soil. Section 2.5 provides some suggestions for predicting soil chemical equilibrium state and section 2.6 discusses the impact of saline-sodic irrigation water on soil structure. The relationships between salinity, sodicity and saturated hydraulic conductivity are presented in section 2.7 and the impact of clay swelling and dispersion on soil pore blockage is discussed in section 2.8.

2.2 Soil structure

2.2.1 Definition and importance

Soil structure refers to the arrangement of soil particles and aggregates and the pore space surrounding them. Soil structure can be expressed as the arrangement of soil aggregates or the distribution of pore sizes. Dexter (1988) describes soil structure as the spatial heterogeneity of different components or properties of soil. This definition is most useful as it includes all characteristics of soil structure that affect root growth. Soil structure has direct effects on many soil properties. These include water retention and conductance, which are dependent on pore space and size. It also influences the root environment of plants through its impact on water, air, moisture availability and soil strength (Marshall and Holmes, 1988). In this way soil structure affects the ability of any soil to support crop growth. Bronick and Lal (2005) recognized the physical aspects of soil fertility in terms of visible forms in soil structure. Pores are arguably the most important physical soil features as they influence most soil processes, including soil biological action. Overall, soil structure affects plant growth by influencing the root distribution and the ability of plants to take up the water and nutrients from soil.

2.2.2 Factors affecting soil structure

Soil structure is influenced by a number of physical and chemical soil properties including amount and type of clay, organic matter, ion composition of soil and the cyclic wetting and drying of soil due to rain and irrigation. Each of these factors is discussed below.

Clay minerals

Clay minerals consist predominantly of layered alumina-silicate (crystalline), although non-crystalline minerals (amorphous) are also an important feature for aggregation in some soils (Powers and Schlesinger, 2002). The ability of clays to incorporate water molecules within their lattice structure allows them to expand and influences the aggregate stability of the soil. The expandability of illitic and smectitic clays can disrupt aggregates during wet and dry cycles (McNeal et al. 1966). The effect depends greatly on the amount of clay and number of shrink-swell cycles (Singer et al. 1992). Considering the physio-chemical features of clays, smectitic clays appear to be more efficient in terms of aggregation than other clays, due to the greater specific surface area, high cation exchange capacity (CEC) and high physio-chemical interaction capacity (Amezketa, 1999). Emerson (1964) found swelling clays are less sensitive to slaking than kaolinite or illite as a result of releasing the developed pressure of entrapped air during the swelling process. However, Le Bissonnais (1996) showed montmorillonite mellowing may occur because of the combination of stress as associated with entrapped air and individual aggregate swelling. Under unstable conditions, such as when the soil solution has a high sodium concentration and/or low electrolyte concentration, soils with high kaolinite content and sesquioxides are more stable than those with high montmorillonite content (McNeal et al. 1966). Amezketa (1999) suggested that the frequent association of kaolinite with iron oxides could be responsible for the high aggregate stability and resistance to slaking.

Clay minerals have both a permanent charge derived from isomorphic substitutions in the mineral structure and a pH-dependent charge. Crystalline particles consist of tetrahedral silica and octahedral aluminium hydroxide in alternating layers as shown in Figure 2.1. Dimorphic clay minerals (1:1), such as kaolinite, are made of silica tetrahedral sheets combined with aluminium octahedral sheets. Generally no isomorphic substitution occurs during formation and thus kaolinite often has no significant permanent negative charge and a small CEC (Matocha, 2002). Trimorphic clay minerals (2:1), such as smectite, are made up of two tetrahedral sheets and one octahedral sheet (Figure 2.1). Due to imperfections in the crystals silicon ions (Si^{4+}) are commonly substituted with aluminium ions (Al^{3+}), and Al^{3+} are substituted with magnesium (Mg^{2+}) ions. Because of this, trimorphic clays have higher permanent charge and CEC than kaolinitic clays. The edge of clay minerals may also exhibit pH-dependent charges that result from protonation and deprotonation of aluminium and silicon groups (Matocha, 2002).

Soils with high content of 2:1 clays are expected to have lower aggregate stability compared to those having high content of 1:1 clay due to swelling property under sodic conditions (Amezketa 1999).

Salts and ions in soil

Soil contains a variety of inorganic and organic salts, some of which are soluble in water and dissociate into ions. Positively charged ions (cations) that are commonly present in soil-water, (referred to as soil solution) and affecting soil structure, include calcium (Ca^{2+}), magnesium (Mg^{2+}), potassium (K^{+}) and sodium (Na^{+}). The double positive charge ions (Ca^{2+} and Mg^{2+}) assist in improving soil structure by forming cationic bridges between soil particles. They also play an important role in decreasing clay dispersion because of their tendency to flocculate clays and organic matter, which assists in enhancing aggregate stability (Zhang and Norton, 2002), although Rengasamy and Marchuk (2011) have shown that Mg^{2+} is only 60% as effective at flocculating clay as compared to Ca^{2+}. Soils developed from iron oxide and clays can also have high aggregate stability as iron present as Fe^{2+} or Fe^{3+} usually makes a stable complex with clay and organic matter.

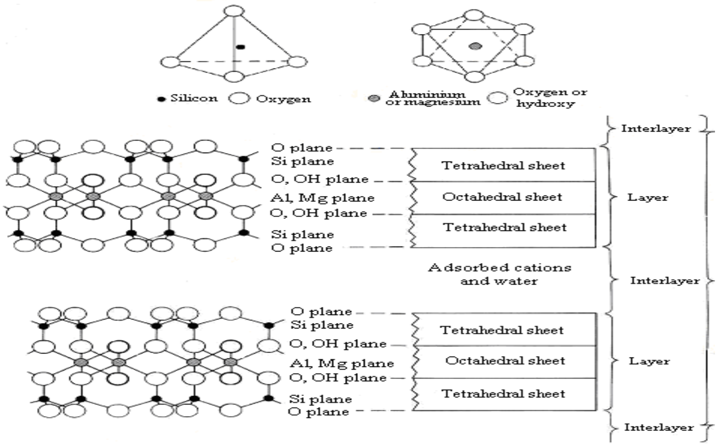
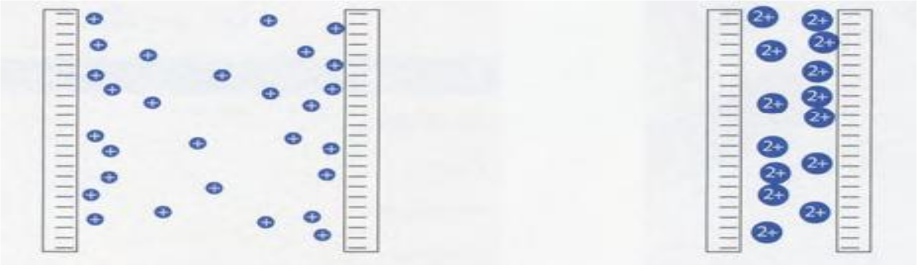


Figure 2-1 Structural components of trimorphic silicate clays (Brady, 1990)

Na^+ ions have a single positive charge, meaning they are less strongly attracted to the active soil surface than polyvalent cations. In addition, the radius of hydrated Na^+ ions is larger than potassium (K^+) ions. While both of these Na^+ characteristics combine to contribute to dispersion, the primary mechanism for increasing the diffuse double layer and instigating dispersion is generally agreed to be the hydrated radius, as two Na^+ ions are required to satisfy the negative charge satisfied by one Ca^{2+} ion. Halliwel et al. (2001) described the behaviour of clay particles in relation to the dominant ions present in the soil solution:

> "*The charge (negative permanent charge) on the clay particles leads to a diffuse double layer formation in which cations cluster to neutralise the surface charge at the interface between surface clay particle and the bulk soil solution. Monovalent cations such as sodium are less attractive than divalent (e.g. calcium) cations resulting in a wider diffuse double layer*"

Osmotic pressure caused by different concentrations of ions in the diffuse double layer relative to the bulk soil solution may act to move the clay particles apart, or closer depending on the electrical conductivity (EC) of the soil solution (Figure 2.2). Dispersion occurs when the individual clay plates move and remain far enough apart to override the van der Waal's attractive forces (Rachad and Dultz, 2007). The thickness of the diffuse double layer may be reduced by introducing cations of high positive charge or small hydrated radius to the soil solution.



Wide diffuse double layer

Narrow diffuse double

Figure 2-2 Effect of sodium and calcium ions on diffuse double layer and clay particle spacing source (Qureshi and Barrett-Lennard, 1998)

Organic matter

There are several positive effects between organic matter and water stable aggregates. Organic matter can play a crucial role in improving and stabilising soil aggregates as it acts as a cementing agent by combining primary soil particles (sand, silt and clay) together into aggregates. Tisdall (1996) found that organic materials such as fine roots assist in stabilising macro-aggregates and thus reduce the surface area from which clay disperses. Hence soil organic matter generally improves soil structure stability as it operates as a cementing agent by joining primary soil particles together into macro-aggregates. Furthermore, Jalali and Ranjbar (2009) evaluated the effect on physical and chemical properties of poultry and sheep manure applied to a sandy loam soil. The application of the sheep and poultry manure increased CEC and the water adsorption of cations such as Ca^{2+}, Mg^{2+} and K^{+} compared to Na^{+} which led to an increase in Na^{+} leaching and improved soil structure.

Wetting and drying

Soils are exposed to seasonal wetting due to rainfall and irrigation and both drainage and between rainfall or irrigation events. When a wet soil dries, negative pressures develop within pores that draw soil particles together causing a decrease in soil volume or shrinkage (Rajaram and Erbach, 1999). Anisotropic shrinkage of soil may lead to the development of a network of cracks and/or surfaces of weakness between soil aggregates (Rajaram and Erbach, 1999). However, rapid wetting of soil may also break large soil aggregates into smaller aggregates due to the release of entrapped air that is known as slaking (Gusli et al. 1994).

2.3 Irrigation water quality

Irrigation water quality is classified according to both the type and total dissolved salts (Ayers & Westcott, 1985). Dissolved salts in irrigation water influence crop production via both direct and indirect effects on the physical and chemical properties of soil, some of which have been discussed earlier. Irrigation water quality is often categorized on the basis of its effects on soil quality and crop yield as: (1) salinity hazard - total soluble salt content, (2) sodium hazard - relative proportion of Na^{+} to a combination of Ca^{2+} and Mg^{2+} ions, (3) acidity, low pH, (4) alkalinity - carbonate and bicarbonate concentration and (5) specific ion toxicity (Bauder et al. 2005).

2.3.1 Salinity hazard

Salinity hazard occurs when salts from irrigation water begin to accumulate within the crop root zone, effectively decreasing the availability of water to plant roots even when the soil is moist. The crop cannot extract sufficient water from the soil due to the increased osmotic pressure of the soil solution thus reducing crop growth and yield. To combat this, plants increase their internal salinity to change the osmotic gradient in their favour and allow adequate water uptake but this can lead to severe toxicities and consequently yield decline, or death (Maas and Hoffman, 1977).

The salinity of irrigation water may be measured as an electrical conductivity (EC_{iw}) or concentration as total dissolved salts (TDS). The EC_{iw} is an indirect measure of dissolved ions present in water and commonly expressed in units of micro-Siemens ($\mu S\ m^{-1}$) or deci-Siemens per metre ($dS\ m^{-1}$). A direct measure of salt concentration, TDS is often expressed in units of milligrams per litre ($mg\ L^{-1}$). Irrigation water with an $EC_{iw} < 0.7\ dS\ m^{-1}$ is generally considered as good quality irrigation water for most crops water while 0.7-3 and > 3 dS m^{-1} are considered as moderate and poor qualities, respectively (Ayers and Westcot, 1985).

The salinity of irrigation water EC can be used as a quick method to estimate salt concentration. The EC increases with increasing the soluble ionic salts and is measured by the electrical current transmitted between two electrodes in the solution. Electrolyte concentration can be estimated from EC for different solutions using:

$$Log\ C^{o} = \dot{a} + \varpi \log EC \qquad 2.1$$

where $C_\circ$ is the salt concentration expressed in $mmol_c$/litre, à and ϖ are empirical parameters which differ with dominant salts in the solutions. This relationship does not always hold, particularly when the solute concentration is too high (Bresler et al. 1982). Bresler et al. (1982) and USSL Staff (1954) suggested a simplified version of equation 2.1 could be used to estimate salt concentration for a range of EC between 0.1 and 10 dS m^{-1}.

$$C^{o} = 10 \times EC \qquad 2.2$$

The salinity of the soil solution also depends on the soil water content at which the salinity measurement is determined. For example, as soil pores dry out the salts remain within solution as the solution volume decreases resulting in an increase in the concentration. In most soils the volume of water extracted from soil samples is usually insufficient to conduct chemical analyses and separating the solution from the solid phase is not easy. Therefore, additional water is usually added to the soil before extracting the solution. Different soil-water extraction ratios, such as 1_{soil}:$5_{water,}$ 1:2.5, 1:1, have been used to measure soil salinity. Saturated soil paste is also used to predict soil-water salinity. In this case the soil solution is extracted after saturation of the soil sample. The water content at saturation is approximately twice that of the field capacity and the salinity concentration measurement, therefore, is nearly half that of the field capacity salinity (Rhoades 1982).

2.3.2 Sodium hazard

While electrical conductivity is an assessment of all soluble salts in water, sodium hazard focuses on sodium, due to its negative effects on soil physical properties. The sodium hazard of water is generally expressed as the sodium adsorption ratio (SAR) and calculated as:

$$SAR = \frac{Na^{+}}{\sqrt{\frac{Ca^{2+} + Mg^{2+}}{2}}} \qquad 2.3$$

where Na^+, Ca^{2+} and Mg^{2+} are the concentrations of sodium, calcium and magnesium ions ($mmol_c$ L^{-1}), respectively. The SAR is an indicator of potential effects of sodic irrigation water on the structural stability of the soil (Ayers and Westcott 1976). Ayers and Westcott (1985) found that irrigation waters with SAR values above 6 reduce soil permeability, particularly when this type of water is applied to clay soils. However, many other factors such as soil texture, organic matter, crop type, climate, irrigation system and management influence the effect of sodium in irrigation water on soil structure (Bauder et al. 2005).

The Exchangeable Sodium Percentage (ESP) is a measure of soil sodicity within the solid phase of a soil and it is defined as the ratio of exchangeable sodium cations to the total cation exchange capacity of the soil (CEC):

$$ESP = \frac{Na^{+} \times 100}{CEC} \qquad 2.4$$

where Na^+ is the ionic concentration of sodium in ($cmol_c\ kg^{-1}$) in soil, and CEC is the cation exchangeable capacity in $cmol_c\ kg^{-1}$. The relationship between SAR and ESP is discussed in section 2.3.5.

2.3.3 Acidity and Alkalinity

Irrigation water can be acidic (pH<7), neutral (pH=7), or alkaline (pH>7), but commonly ranges between 7 and 8. Irrigation waters exceeding a pH of 8.5 generally have high concentrations of carbonate and bicarbonate that may exacerbate sodicity effects in soil. Carbonate (CO_3^{2-}) and bicarbonate (HCO_3^-) ions react with soluble Ca^{2+} and Mg^{2+} in irrigation water or in the soil during drying to form insoluble lime, leaving sodium as the dominant ion in solution and increasing SAR significantly (Bauder et al. 2005). Irrigation water with high Residual Sodium Carbonate could increase pH, electrolyte conductivity and SAR of saturated soil paste extracts (Prasad et al. 2001). The bicarbonate hazard of irrigation water is often expressed as the RSC:

$$RSC = (HCO_3^- + CO_3^-) - (Ca^{2+} + Mg^{2+}) \qquad 2.5$$

where the concentrations of ions are expressed in $mmol_c\ L^{-1}$. In general, irrigation water with RSC < 1.25 $mmol_c\ L^{-1}$ is considered safe, between 1.25 and 2.5 $mmol_c\ L^{-1}$ as marginal, and ≥2.5 $mmol_c\ L^{-1}$ as unsuitable for irrigation, unless it is treated with gypsum (Nishanthiny et al. 2010; Harivandi, 1999). Ghiberto et al. (2007) studied the influence of residual sodium carbonate at values greater than 1.25 $mmol_c\ L^{-1}$ on the hydraulic conductivity of a mollisol and found that the hydraulic conductivity decreased by 10% and 79%, and ESP increased from 1 to 10 for 0-7 cm and 15-25 cm soil depth, respectively.

2.3.4 Specific ion toxicity

Specific ions may be toxic to crop plants at high concentration. These ions are usually trace minerals, including Cl, B and Na some of which are essential for plant growth, but required at low concentrations (Ayers and Westcott, 1985). Some of these elements may also interfere with the availability of other nutrients and cause toxicity when their concentrations exceed the tolerance limit of the crop (Bauder et al. 2005). Ion toxicity effects are usually worse when water is applied with sprinkler irrigation causing leaf burn (Bauder et al. 2005). In general, ion toxicity problems affect crop growth directly, whereas salinity and sodicity generally affects crop growth indirectly via soil physical degradation or osmotic effects on water uptake.

2.3.5 Relationship between SAR and ESP

Sodium exchangeable percentage is quite commonly used as an indicator of sodicity. However, Qadir and Schubert (2002) identified that ESP may not always accurately identity soil sodicity risk due to: (1) in calcareous soils $CaCO_3$ and $MgCO_3$ may dissolve during the extraction of Mg^{2+} and Ca^{2+} causing an increase in CEC; (2) the extraction may lead to Na^+ removal from a sources that are not truly form of exchangeable sodium for example sodium zeolites; (3) in variable charge soils the CEC depends on soil pH. Chartres (1993) suggested that the SAR may be more suitable for predicting soil sodicity as it estimates the activity of a range of cations in solution. there are also less parameters required for measurement of the SAR (the concentration of Ca^{2+}, Mg^{2+} and Na^+) and extracted soil-water used for measuring the EC can be used to determine the concentration of these elements (Qadir and Schubert 2002). However, Ayers and Westcott (1985) found that the SAR does not consider the changes of Ca^{2+} concentration in soil solution resulting from the change in solubility of Ca^{2+}. In addition, some errors in SAR calculation may occur following irrigation in which Ca^{2+} concentration may increase due to dissolution of minerals or decrease due to precipitation as carbonates, bicarbonates or sulphates.

There is a close relationship between the equilibrium ESP of the soil and the SAR of the irrigation water applied. United State Salinity Laboratory Staff (1954) found that the ESP of the soil is related to the SAR of irrigation water (in the SAR range of 0-40) when in equilibrium:

$$ESP = \frac{100\,(-0.0126+0.01475\,SAR)}{1+(-0.0126+0.01475\,SAR)} \qquad 2.6$$

However, it is important to note that the relationship between ESP and SAR is different from one soil to another due to differences in soil texture and the concentration of clay mineral type. The comparison of different empirical parameters for equation 2.6 for a range of SAR is shown in Figure 2.3. This figure shows some equations lead to negative ESP values at low values of SAR, which is not a practical possibility. The United States Salinity Laboratory Staff (1954) also note that the developed equation provides negative ESP values when the SAR is below 0.5.

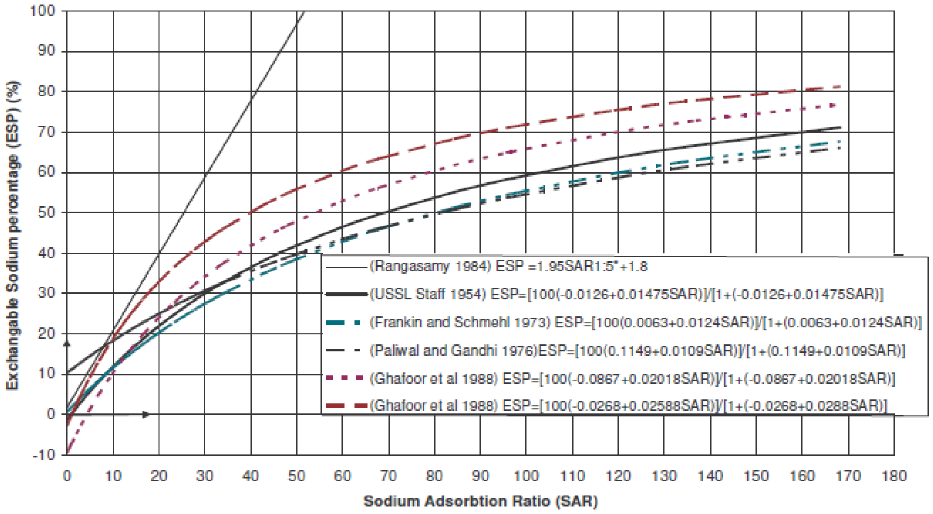


Figure 2-3 Relationships between SAR-ESP from Qadir and Schubert (2002)

2.4 Ion exchange processes

Ion exchange in soils is a reversible process in which cations and anions are exchanged between solid and liquid phases or solid and solid phases (if in close proximity to one another) (Sposito 1989). Ion exchange processes involve the adsorption of ions onto colloidal surfaces and desorption of ions from these surfaces (Toth 1964). Ion exchange influences various soil characteristics and behaviour, including swelling and shrinkage, leaching of electrolytes, weathering of minerals and adsorption of nutrients by plants (Wiklander 1964) and occurs almost entirely in the clay and silt fractions, as well as within the organic fraction (Valzano 2000).

Boyd et al. (1947) found that ion exchange is diffusion controlled and the reaction rate is limited by mass-transfer phenomenons that are either film diffusion (FD) or particle diffusion (PD) controlled. Aharoni and Spark (1991) describe the specific transport processes in a soil-solution system in six phases (Figure 2.4) including (1) transport in the soil solution, (2) transport across a liquid film at the particles and liquid interface (FD) (3) transport in liquid filled macropores (PD), (4) diffusion of a sorbate at the surface of the solid surface (PD) (5) diffusion of a sorbate occluded in a micropore (PD) and (6) diffusion in the bulk of the solid.

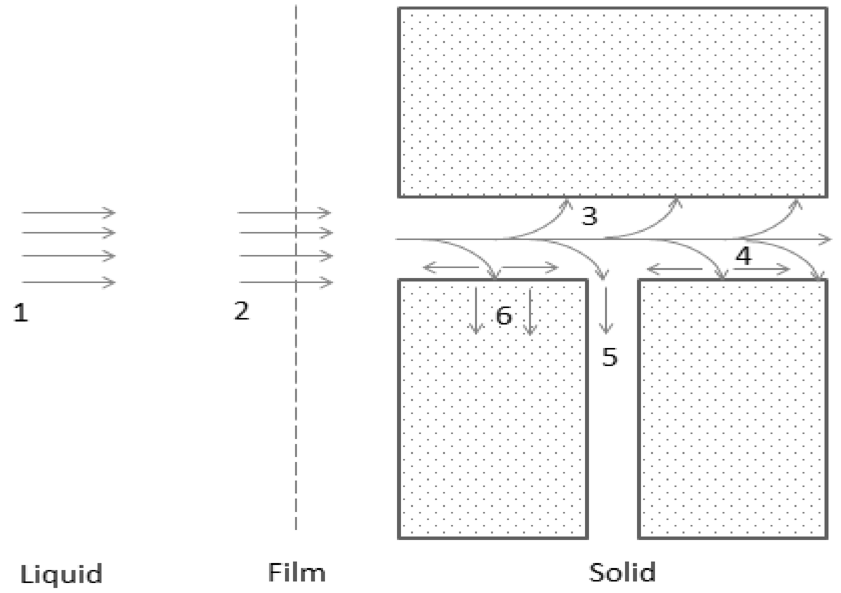


Figure 2-4 Transport processes in solid-liquid soil reactions (Aharoni and Sparks 1991)

Tang and Sparks (1993) noted that the ionic soil chemical reactions are very fast and occur within millisecond time scales. The rate of these reactions is affected by clay type (Sparks 2003).

Clay mineralogy is one of the most important factors influencing the rate of ion exchange in soils (Sparks 1988). Sorption reactions on kaolinite and smectite clays are usually faster than on vermiculite and micaceous minerals due to the variations in physical structures of clays (Sparks 2003). Jardine and Sparks (1984) found the rate of ion exchange processes on kaolinite and smectite clays are usually rapid to due to strong hydrogen bonding between adjacent tetrahedral clay sheets. This permits rapid exchange on the external sites (Sparks 1988). Weaker bonds between sheets in smectite allow ions present in the soil solution to enter the interlayer space, encouraging rapid exchange since ions in solution access the exchange sites on the surface of clay sheets (Sparks 2003).

The ion exchange rate is often slow for vermiculite and micas that have multiple (e.g. planar, edge and interlayer sites) exchange sites (Sparks 2003). Comans and Hockley (1992) suggested three different exchange rates in these clays with high exchange rates on external sites, intermediate exchange rates on edge sites and low exchange rates on interlayer sites. Sparks (2003) explained that low exchange rates on interlayer sites are proof of partial to total collapse of the space between clay sheets and as a result, sorption and desorption processes in this space are interparticle diffusion and mass transfer controlled. This suggests that, the processes of ion exchange within soil are influenced by soil pore size and soil clay type. However, no studies have been found dealing with the effect of soil pore size on ion exchange rate in relation to changes in soil K_S when saline-sodic irrigation water is applied to soil with different clay content and type.

2.5 Equilibrium prediction

Shackelford and Redmond (1995) studied the factors influencing the EC breakthrough from soil columns as an indicator of chemical equilibrium between the leached and applied solutions. Electrical conductivity breakthrough occurs when EC of the leachate is normalized with respect to EC of the applied solution (Shackelford 1999). Results (Figure 2.5) comparing measured (Shackelford and Redmond 1995) and predicted (Shackelford et al. 1999) EC breakthrough curves showed that estimates of breakthrough curve ranged from good to excellent compared to measured breakthrough. Further, the results indicate that chemical equilibrium cannot be achieved before complete EC breakthrough is reached.

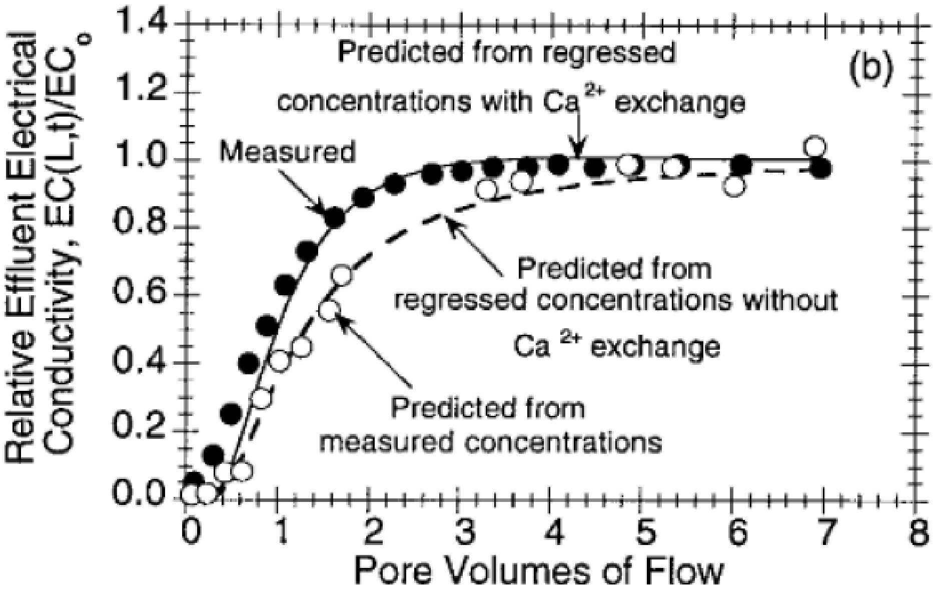


Figure 2-5 Measured and predicted EC breakthrough curves (Shackelford et al. 1999)

Other studies (Misra and Sivongxay 2008 and Jalali and Ranjbar 2009) investigated EC breakthrough curves and exchangeable cations measurements (Wieck 2012) in flow-pass experiments demonstrate that the EC steady state may be approached within the first three pore volumes (PV) during the leaching. However, exchangeable cation measurements showed that the equilibrium state was not reached after applying more than 10 PV of solution (Misra and Sivongxay 2008; and Jalali and Ranjbar 2009; Wieck 2012).

2.6 Impact of saline-sodic irrigation water on soil structure

Stable soil aggregates are important in maintaining adequate hydraulic conductivity that is beneficial for plant growth. Maintaining K_s of the soil ensures movement and storage of irrigation water in the soil and its availability to plants. In addition, an adequate soil K_s is required for the redistribution of applied irrigation water within the root zone.

The application of high Na^+ concentration irrigation water may increase the soil ESP and that may cause clay swelling and dispersion that results in reductions in soil K_s values. This is because Na^+ has a relatively large size, single electrolyte charge and hydration states that tend to cause physical separation of soil particles. The physical separation of soil particles results in sufficient distance between individual soil particles such that repulsive forces between like molecules exceed bonding forces and dispersion occurs (Ayers and Westcot, 1976). A second, somewhat reversible process associated with increasing soil ESP is platelet and aggregate swelling. Smaller, non-hydrated divalent cations such as Ca^{2+} do not have this same effect because they tend to cluster closer to the clay particle (their second positive charge causes a stronger attraction to clay surface than Na^+, which has a single charge). Basically, attractive forces which bind clay particles together are disrupted when too many Na^+ ions enter between the clay particles. When such separation occurs, repulsive forces begin to dominate, and the soil disperses (Buckman and Brady, 1967; Hanson et al. 1999).

Quirk (2001) described the processes using a “three plate model” in which individual clay crystals overlap as (Figure 2.4). This model is useful for showing the swelling and dispersion processes and the effect of ESP on clay dispersion. When water or an electrolyte solution is applied to soil, repulsive pressure (P_R) associated with the osmotic effect and changes in the diffuse double layer develop over the surface area of large slit-shaped pores. However, an attractive pressure (P_A) associated with Van Der Waals forces operate over the surface area of the more closely aligned crystals (Kjellander et al. 1988). Dispersive cations mainly Na^+ tend to concentrate in the slit shaped pores (Sumner, 1993) and form extensive double layers (compared to smaller double layer for cations of higher valence).

Many researchers (McNeal and Coleman 1966; Shaingberg et al 1981; Manneer et al 2001; Misra and Sivonggxay 2009) have studied the effect of using saline-sodic irrigation water on changes in soil K_s values. Oster’s (1994) review indicated that soil dispersion and slaking is more pronounced when the applied water has a high SAR and low salinity, due to increased ESP values on the active clay surfaces. Manneer et al. (2001) examined the effects of different SAR irrigation water (SAR between 5 to 17) on the permeability of two New Zealand soils: an Allophanic and a Gley soil. These authors found the ESP increased at several depths in each soil. Moreover, in their laboratory studies, there was a reduction in soil K_s values with SAR > 3.5 and 8.5, for the Allophanic and Gley soils respectively. The reduction in soil K_s in this case was related to a release of dissolved organic matter leading to changes in soil physical properties.

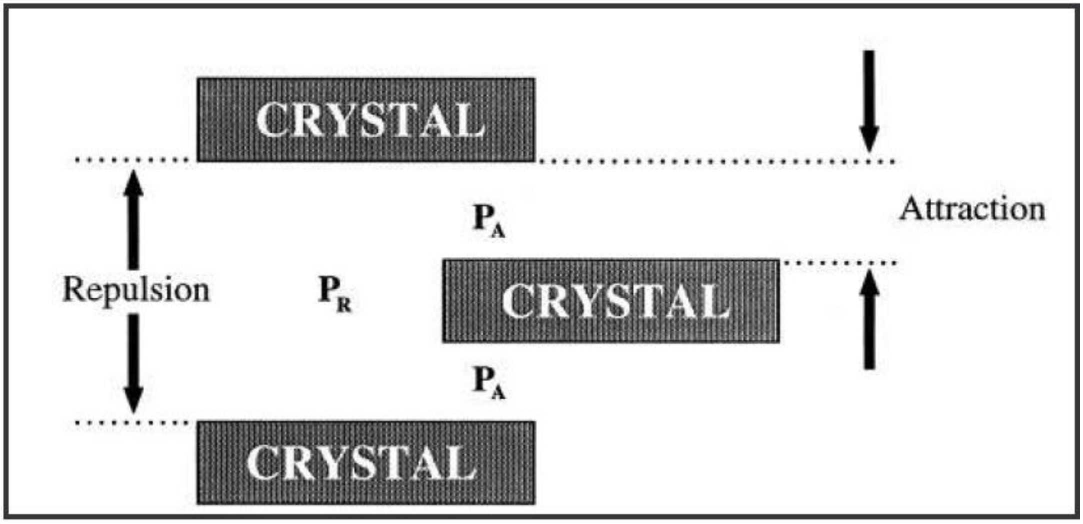


Figure 2-6 A simple 3-plane model to describe the arrangement of clay crystals in clay crystals in clay domain from Quirk (2001)

When saline-sodic water is applied to clay soils, soil slaking, dispersion and reduction in soil K_s depend greatly on the clay type. For example, swelling and dispersion of smectite dominated clay soils caused movement of clay fractions and decreasing soil K_s when the soils were leached with low salinity water (Shainberg et al. 1981; McNeal and Coleman, 1966). Goncalves et al. (2006) studied the effects of irrigation water with SAR values of 10.4 and 19.6 on the saturated hydraulic conductivity of two soils and found a significant decrease in soil K_s of the two soils. It was suggested that these reductions in soil K_s were related to changes in macro and micro porosity changes by dispersed clay.

Clearly, many studies that examine the effect of dispersed clay on soil pore blockage have used K_s reductions as an indicator of soil pore blockage. However, there is a lack of studies that investigate the impact of saline-sodic irrigation water on changes in soil pore network.

2.7 Relationships between salinity, sodicity and saturated hydraulic conductivity

Cook et al. (2006) noted that structural stability of soil depends on the interaction between soil sodicity and salt concentration in the soil solution. Clay swelling and dispersion occur at a certain relative ESP when the salt concentration in soil the solution is less than a critical electrolyte concentration which is referred to (Quirk and Schofield 1955) as the threshold electrolyte concentration (TEC). Threshold concentration is defined as the salt concentration at which the soil K_s begins to decrease for a certain sodicity value (Quirk and Schofield 1955).

The reduction in soil K_s associated with changes in water quality is a progressive process. Thus, Quirk and Schofield (1955) arbitrarily suggested that the critical reduction in soil K_s for the TEC by 15% from the stable soil K_s. However, McNeal and Colman (1966) suggested using 25% reduction as the critical reduction in soil K_s associated with the TEC for some American soils. This may be due to differences in clay minerals types (Figure 2.7).

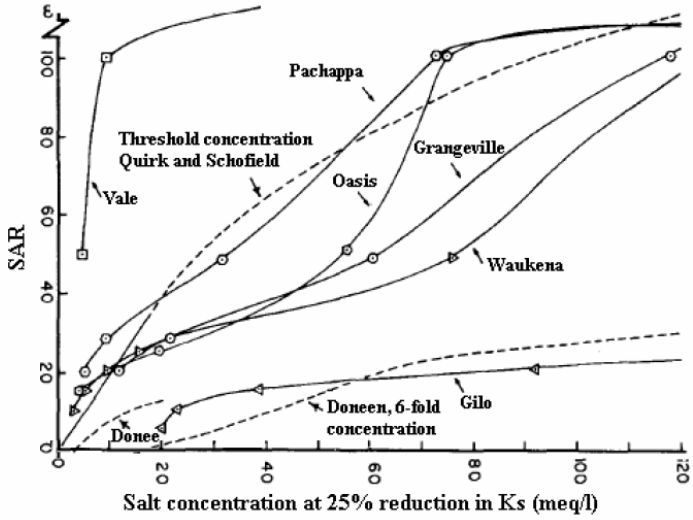


Figure 2-7 combinations of salt concentration and SAR at which a 25% reduction in soil Ks occurred (McNeal and Coleman 1966)

More recently, Bennett and Raine (2012) used a 20% reduction in soil K_S as the critical threshold and found that even soils within similar order can have considerably different TEC curves due to differences in clay content and type and organic matter content (Figure 2.8).

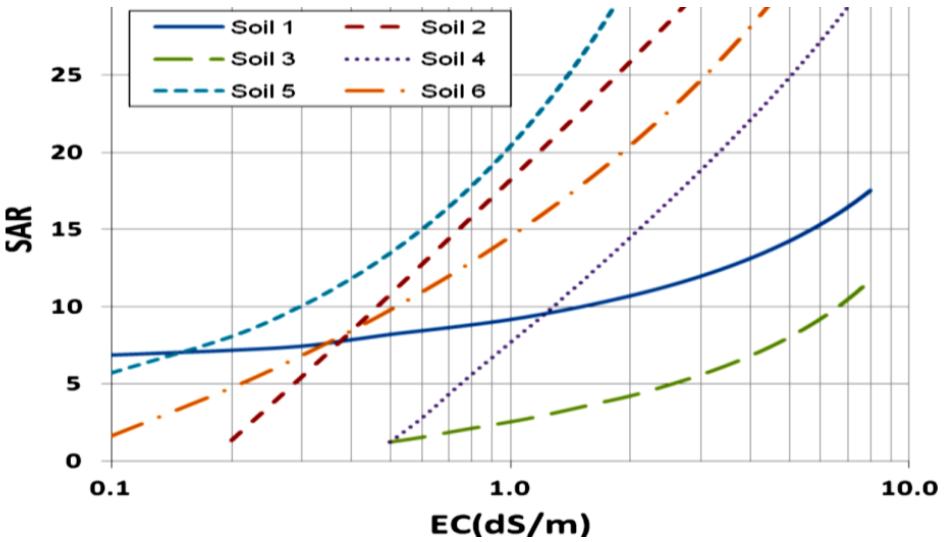


Figure 2-8 Comparison of the TEC (20% reduction in soil Ks) curves for six soils where soil 1, 2 and 6 are Vertosol and 3, 4 and 5 are Chromosols (Bennett and Raine 2012)

2.8 Clay swelling, dispersion and soil pore blockage

The potential for soil pore blockage and soil K_S reductions are often estimated (as discussed in section 2.2) on the basis of the salinity and relative Na^+ content of irrigation water applied. Soil aggregates slaking, clay swelling and dispersion are the main processes resulting in reduction in soil pore size (Quirk and Schofield 1955; Levy 2005). Slaking occurs when water is applied to dry clay soil. Auerswald (1995) found that the release of entrapped air from soil pores was the major cause of aggregate degradation of pre-wetted aggregates for 100 arable soils during percolation. Abu-Sharar et al. (1987) suggested that the degree of slaking depends on the SAR of the applied water, and for soils with low EC values, swelling then dispersion can occur during the last stages of slaking. However, reductions in soil K_s due to soil pore blockage are largely attributed to clay swelling and dispersion (Levy et al. 2005). Frenkel et al (1978) stated that clay swelling reduces soil pore sizes and clay dispersion blocks soil pores. However, reductions in soil K_s resulting from clay dispersion and clay sediments movement are permanent while soil K_s changes due to swelling is a reversible processes (Frenkel et al. 1978).

Reductions in soil pore size when irrigation water with high SAR and salt concentrations is applied could result from clay swelling. Schofield (1955), McNeal and Coleman (1966) and Russo and Bresler (1977) suggested that clay swelling associated with increasing SAR values may result in either complete or incomplete blockage of soil pores. Quirk (2001) stated that clay swelling under sodic conditions might be extremely dispersive because osmotic swelling forces draw water into the interlayer space between clay platelets pushing clay particles apart and leading to breakdown of soil aggregates.

Under sodic conditions, clay swelling has been noted at ESPs below 15% (Oster et al. 1980). Conversely, Hanson et al. (1999) suggested that in soils with a high content of 2:1 clays swelling processes occur as soil comes into contact with water. Frenkel et al. (1978) studied the changes in water potential within soil cores dominated by montmorillonite, vermiculite and kaolinite and treated with low SAR and EC water. The results showed that the upper parts of the soil cores were blocked due to clay sediment movement. Emerson and Bakker (1973) identified that when low SAR and EC solution is applied to soil, the EC of the soil solution within the macropores may reduce to values less than the critical concentration. In this case, a sharp osmotic gradient will be generated between the solutions within micropores and the solution in the macropores. This osmotic gradient will draw water into the micropores leading to clay dispersion of the outer soil aggregate layers. As the process continues, complete degradation of soil aggregates may occur (Jayawardane and Beattie 1978).

Clay dispersion processes are greatly dependent on the osmotic gradient created between the soil solution infiltrating within the macropores and the soil solution in the micropores. Keren and Singer (1988) showed that applying distilled water to soil cores after leaching with a 10 $\mathrm{mmol_c\ L^{-1}}$ solution having SAR 5, 10 or 20 resulting in severe reductions in soil K_s and the presence of dispersed clay in the leachate.

It is clear that clay swelling and dispersion are a major cause of soil pore blockage and soil K_s reductions. However, there is a need to better understand the mechanisms of soil pore blockage and the impact of soil pore size on the potential of soil pore blockage. While literature concerned with soil compaction, clay dispersion and soil structural integrity has considered pore blockage mechanisms to some extent, this work has only been conducted on a limited range of soils types (Red Ferrosol, Grey Vertosol and Black Vertosol) that commonly found in Queensland region and not using the dominant soils exposed to saline-sodic groundwater or coal seam gas associated water in southern Queensland. Therefore, the mechanisms of soil pore blockage and the potential effect of soil pore size on dispersed clay leaching and accumulation with depth will be investigated in this study for soils with different clay types under sodic conditions.

2.9 Conclusion

Soil structure has a direct impact on water and solute movement within the soil profile. Degradation of soil structure can produce complex problems related to waterlogging and aeration that may affect crop growth and productivity by restricting root distribution and the crop ability to take up water and nutrients.

The use of saline-sodic irrigation water can increase soil ESP by exchanging Na^+ in the applied water and cations on the soil exchange surface. These exchange processes are affected by clay mineralogy and soil pore geometry. Increasing soil ESP may lead to soil structural degradation due to clay swelling and dispersion. This occurs because Na^+ has a relatively large ionic size, a single electrolyte charge and hydration states that tend to separate clay particles.

The potential for soil structural degradation depends on the interaction between soil sodicity and salt concentration in the soil solution. Clay swelling and dispersion occurs at a certain relative ESP when salt concentration in the soil solution is less than a critical electrolyte concentration. When low salinity water is applied, this is due to the osmotic gradient created between the applied solution equilibrating within the macropores and the soil solution in the micropores. The sharp osmotic gradient generated between micropores and macropores draws water into the micropores leading to clay dispersion of the outer layers of soil aggregates. Clay swelling and dispersion are the main processes resulting in a reduction in soil K_s through both reductions in soil pore size and through pore blockage with dispersed clay.

This literature review has shown that the effect of using saline-sodic irrigation water on clay swelling, dispersion and reductions in soil K_s has been widely documented. Most of these studies have considered the reduction in soil K_s as an indicator of clay dispersion and soil pore blockage. However, few studies have considered the impact of the differences in soil pore size on exchange processes, dispersed clay movement and the potential for soil pore blockage when saline-sodic irrigation water is applied. Therefore, this research will investigate the effect of soil pore size on cation exchange processes, dispersed clay movement and the mechanisms of soil pore blockage in relation to changes in soil K_s. These processes will be studied in re-packed soil column experiments using saline-sodic water applications to simulate structural changes in cultivated surface soils irrigated with marginal quality water.

Chapter 3: General Methodology

3.1 Introduction

The work reported in this thesis consists of a number of discrete experiments. Although each experiment was a separate entity, many of the materials and methods used in these experiments were similar. To avoid unnecessary repetition, the common methodologies are described in this chapter. Specific details for each experiment are described in the relevant chapters.

3.2 Soil selection and sampling

Three different soils (Table 3.1) were selected to examine the effect of saline-sodic irrigation water with different SAR values on the rate of ion exchange, dispersed clay movement and soil pore blockage. The soils selected were a Black Vertosol (BV), Grey Vertosol (GV) and Red Ferrosol (RF) which are all commonly found in area of Queensland where saline-sodic groundwater or water from coal seam gas (CSG) operations is being applied to land.

The GV and BV soils had been previously collected from surface (0 – 10 cm) layers and obtained for this study from the National Centre for Engineering in Agriculture (NCEA) soil library. The RF soil was collected (Figure 3.1) from the 0 – 10 cm depth from a paddock at the University of Southern Queensland farm (27^{o} 36$^{'}$ 36$^{''}$ S, 151^{o} 55$^{'}$ 53$^{''}$ E, 690 m elevation). The RF soil is well structured soil with a high content of free iron oxides in the B2 horizon (CSIRO 2007). The BV soil is a heavy clay fine structured soil with a high content of 2:1 clay. It has a tendency to swell when wet and develops cracks when dry (CSIRO 2007). This soil was collected from a dairy farm paddock at Yalangur, Queensland (26^{o} 23; 57$^{''}$ S, 151^{o} 49$^{'}$ 26$^{''}$ E, 429 m elevation). The GV soil has similar properties to BV soil but with less swelling and shrinking as it contains a smaller proportion of 2:1 clay. The GV soil was collected from at farm a Roma in southern Queensland (25. 5020 S; 149.5713 E; 200 m). Following collection, the soils were dried by spreading out on a plastic sheet to a depth of less than 3 cm in the laboratory. After 10 days, the soils were crushed by hand to pass through a 2 mm sieve. After crushing the soils were kept sealed in plastic bags.

3.3 Methods

3.3.1 Soil core preparation

A 20 kg sample of each soil was spread on steel trays and 3.5 L of tap water was sprinkled on the top surface of each soil. After 3 days, the soils were homogeneously mixed and sealed in a plastic bag over night to equilibrate. The gravimetric moisture content of the soil was measured (3 replicates) as 18 (± 0.5), 21 (± 1) and 24 % (± 0.8) for RF, GV and BV soil, respectively. The soils were stored in sealed plastic containers until used for packing the cores.

Table 3-1 Selected chemical and physical properties of the three soils

Properties	*Unit*	*Red Ferrosol*	*Grey Vertosol*	*Black Vertosol*
			Mean value ± SE	
EC (1:5 soil – water ratio)	dS m^{-1}	0.55 ± 0.01	0.72 ± 0.008	0.98 ± 0.02
pH (1:5 soil - water ratio)		5.30 ± 0.05	7.55 ± 0.01	7.49 ± 0.04
Clay (<0.002 mm)	%	38.15 ± 0.25	42.70 ± 1.8	69.15 ± 1.35
Silt (0.02-0.002 mm)	%	27.15 ± 0.55	41.05 ± 2.75	17.10 ± 2.1
Sand (2.0-0.02 mm)	%	34.70 ± 0.8	16.25 ± 0.95	13.75 ± 0.75
Total Na concentration	mg kg^{-1}	120 ± 10	400 ± 21	450 ± 12
Total Ca concentration	mg kg^{-1}	2108 ± 81	3594 ±206	4813 ± 189
Total Mg concentration	mg kg^{-1}	900 ± 51	2090 ± 216	2892 ± 306
Total K concentration	mg Kg^{-1}	380 ± 21	650 ± 33	1200 ± 71
Exchangeable Na concentration	meq 100 g^{-1}	0.42 ± 0.001	1.3 ± 0.004	1.45 ± 0.001
Exchangeable Ca concentration	meq 100 g^{-1}	9.50 ± 0.57	17.24 ± 1.0	21.91 ± 1.1
Exchangeable Mg concentration	meq 100 g^{-1}	7.10 ± 0.3	15.40 ± 0.8	20.21 ± 1
Exchangeable K concentration	meq 100 g^{-1}	0.88 ± 0.001	1.35 ± 0.05	2.94 ± 0.008
Cation exchange capacity (CEC)	meq 100 g^{-1} soil	18.99 ± 1.1	36.44 ± 1.6	48 ± 2
Exchangeable sodium percentage (ESP)	%	2.21 ± 0.003	3.56 ± 0.02	3.01 ± 0.09

Figure 3-1 Google image of University of Southern Queensland farm showing where the RF sample was taken

Plastic PVC tubes (50 mm internal diameter and 80 mm high) were used as soil cores. The soil was packed in the tubes at the initial desired soil ρ_b based on dry mass for each experimental chapters where 8/9 of the soil was packed from the top and then the core was turned upside down after dropping 4 times from 2 cm height to obtain uniform packing then add the other 1/9 from the other side following the method described by Misra and Li (1996). The initial soil ρ_b was calculated from the relationship between the core volume and soil mass. It was assumed that leakage along the cylinder walls would be non-significant (Reynolds et al 2000) as clay soil was used and the packing insured good contact between the soil and cylinder wall. The cores were supported with cheesecloth from below and a fast filter paper was placed on the top to prevent soil surface disruption during hydraulic conductivity measurement. Rims of 15 mm height were attached to the top of the cores with a water resistant seal to provide the water head during leaching.

3.3.2 Solution preparation

A combination of calcium and sodium chloride salts were dissolved in distilled water to make solutions of different SAR and EC. Solutions were prepared to simulate (a) good quality (SAR = 0.11 and EC = 1 dS m^{-1}) irrigation water (GQW), (b) saline-sodic water (SAR = 10 and EC = 2 dS m^{-1}) low quality water similar to that often used in agriculture, and (c) highly saline-sodic water (SAR = 50 and EC = 2 dS m^{-1}) similar to that produced during coal seam gas extraction and treatment.
The EC of the solutions was checked by taking three replicates of 100 cm^3 from each container and measuring using a calibrated EC meter (LabCHEM- CP-Cond/pH, version 1.01).. Results are shown in Table 3.2.

Table 3-2 Mass of salt dissolved in 20 L of distilled water to create the water quality solutions

Water quality	Target		Measured EC (dS m^{-1})	Mass (g)	
	SAR	EC (dS m^{-1})		$CaCl_2$	NaCl
GQW	0.1	1	1.05 ± 0.02	14.34	0.28
Saline-sodic water	10	2	2.1 ± 0.05	6.89	17.9
CSG water	50	2	2.05 ± 0.02	0.45	23.0

3.3.3 Gravimetric water content measurement

The gravimetric water content of soil samples was measured throughout this work following the method of Hesse (1971). 10 g of each soil was weighed using an electronic scale (OHAUS corporation, USA, d = 0.1 mg) in aluminium containers of known weight. The soil samples were dried in an oven at 105 C° for 24 hours. The soil samples were then taken out of the oven, covered with the lids and then left for 30 minutes to cool. The samples were then weighed and the gravimetric water content calculated using:

$$Gravimetric\ water\ content\% = \frac{Soil\ wet\ weight - soil\ dry\ weight}{soil\ dry\ weight} \times 100 \qquad 3.1$$

3.3.4 Particle size analyses

Particle size analyses were conducted using a method similar to that suggested by Raine and So (1994). 20 g of each soil (dry soil) was weighed in 100 cm^3 plastic containers (W1). Ultrasonic energy was applied using a Branson 250 ultrasonic probe at 65 % power for 10 minutes. The suspension was carefully transferred to 1000 cm^3 cylinders by washing with distilled water and the cylinders were then made up to 1000 cm^3. The cylinders were left on the laboratory bench to equilibrate at 25 C°. The suspensions were then mixed thoroughly and left to settle for 1 minute and 48 seconds. A side entry pipette was then used to extract a 10 cm^3 aliquot from the suspension at a depth of 100 mm from the surface. The aliquot was then transferred to an aluminium container of known weight and dried in an oven for 24 hours at 105 C°. The difference in weight before and after drying (W2) was calculated and used to calculate the percentage of the clay + silt
(i.e. < 20 um fractions) according :

$$Silt\ and\ clay\% = \frac{\left(\frac{w_2}{10}\right)\times 1000}{w_1} \times 100 \qquad 3.2$$

The suspensions were stirred again and left to settle for 1 hour and 35 minutes before extracting 10 cm^3 from the suspension at a depth of 20 mm from the top. The aliquot was dried as above and used to calculate the clay percentage. The sand and silt percentages were then calculated as:

$$Silt\% = (clay + silt\%) - clay\% \qquad 3.3$$

$$Sand\% = 100 - (clay\% + silt\%) \qquad 3.4$$

3.3.5 Cation exchange capacity (CEC)

Cation exchange capacity was measured using the Richards (1954) method. 5 g of air dry soil was weighed, transferred to a centrifuge tube (45 cm^3) and 33 cm^3 of trihydrate sodium acetate was added. The tubes were shaken by a shaker (John Morris Pty. Ltd, Model No 3521, Illinois USA) for 10 minutes at 100 r p m and then placed in a centrifuge (Model No GT-7020B, Austalia) for 15 minutes at 3000 r p m. The supernatant was removed and the process repeated 3 times. 33 cm^3 of 95 % ethanol was then added to the soil samples and shaken for 5 minutes, centrifuged for 15 minutes, and the solution drained. This step was repeated until the EC of the solution was less than 400 µS/ cm to ensure the removal of soluble cations. 33 cm^3 of 1N ammonium acetate was then added 3 times to the samples, shaken and centrifuged to replace the adsorbed sodium on the active soil surface with ammonium. In each case the solution was transferred to a 100 cm^3 beaker and after the third extraction 1 cm^3 of 1N ammonium acetate (1N) was added to complete the volume to 100 cm^3. The sodium concentration in the extraction solution was measured using an Atomic Absorption Spectrometer (AA 7000, Japan). The following formula was used to calculate cation exchange capacity (CEC):

$$CEC\ (cmol\ kg^{-1}) = Na\ (cmol\ kg^{-1}) \times \frac{A}{wt} \times \frac{100}{1000} \qquad 3.5$$

3.3.6 Soluble and exchangeable cation concentrations in soil

Soluble and exchangeable cations were measured using the method suggested by Rayment and Higginson (1992). For the soluble ion measurement, 5 g of air dried soil was weighed, transferred to a centrifuge tube (45 cm^3) with 25 cm^3 of distilled water and shaken by a shaker (John Morris Pty. Ltd, Model No 3521, Illinois USA) for 10 minutes at 100 r p m. The tubes were then placed in the centrifuge (Model No, GT-7020B, Australia) for 15 minutes at 3000 r p m and then the supernatant was transferred to a 100 cm^3 container. The solution was diluted with distilled water at either 1: 1500 for sodium measurement or 1:500 for measurement of the other cations (calcium, potassium and magnesium). Atomic absorption was used to measure the concentration of the soluble cations in the diluted solutions.

To measure the exchangeable cations, 5 g of air dried soil was weighed, placed into centrifuge tubes, and washed with 95 % ethanol twice and then 25 cm^3 of 1M ammonium chloride was added. The tubes were shaken for 10 minutes and centrifuged for 15 minutes. The supernatant was then transferred to 100 cm^3 containers and diluted with distilled water at either 1:2000 for sodium measurement or 1:800 for measurement of the other cations (calcium, potassium and magnesium).

The atomic absorption spectrometer was used to measure the concentration of exchangeable cations (Na^+, Ca^{2+}, Mg^{2+} and K^+) in the diluted solutions.

3.3.7 Soil impregnation and soil section preparation

The Murphy (1986) method was used to create resin impregnated soil core samples. The cores for each soil were dried above a bath of acetone. 500 cm^3 of acetone was poured in a glass container with a mesh attached above the acetone level. The soil cores were placed above the mesh and the lid was fixed on the top of the container. The lid was removed daily for 2 hours during the day and 2 hours at night to reduce any effect of the acetone on the PVC cores. The acetone was replaced every three days until the soil surface was dried. This took 15 days for the Red Ferrosol and 27 and 41 days for the Grey Vertosol and the Black Vertosol, respectively. After drying, the cores were saturated from beneath with a mixture of 30% polyester resin (Poly Plex Clear Casting Resin, Fibre Glass International Division, Nuplex Industries, Australia), 65% styrene monomer (Fibre Glass International Division, Nuplex Industries, Australia) and 5% methyl ethyl ketone peroxide catalyst (Fibre Glass International Division, Nuplex Industries, Australia). The cores were left to dry for 2 weeks under laboratory conditions. Once dry, the soil was taken out of the PVC pipes and impregnated for 24 h under vacuum in a mixture of 40% polyester resin, 50% catalyst, 5% green yellow fluorescent dye (Solvent Fluorescent 43, All Colour Supplies Pty Ltd) and 5% hardener. The soil was left to cure for 7 weeks and was then sectioned with a domain saw into 2 cm horizontal cross-sections. The surface of the cross-sections were smoothed using 400 grit sand paper attached to a sander (Buehler, USA) at 285 r p m. Once the surface of the soil section was flat, the same instrument (Buehler, USA) was used to wet polish the section surface in the presence of water at a lower speed (190 r p m). The soil section was then placed under the microscope set at a magnification of 65 X to obtain high-quality resolution. The images were captured and analysed using a custom made program (prepared by NCEA staff) developed to measure the total porosity and pore size distribution based on colour separation. The yellow and green areas represented the impregnated pores (filled with polyester resin) and the other colours represent the soil matrix.

3.3.8 Statistical analyses

Experiments in this study were conducted in a completely randomized design. Statistical analyses were undertaken using the Statistical package for the Social Sciences (SPSS) v10 for windows 7 (Cramer, 2004). Analyses involved one and two way ANOVA to evaluate the effect of the interaction between parameters studied. Least significant difference (LSD) was used to compare the means at a probability level of 5%.

Chapter 4: Influence of Suspended Clay Concentration in Infiltration Water on Pore Blockage and Saturated Hydraulic Conductivity

4.1 Introduction

Soil pores can be considered as a group of connected voids distributed within the soil and through which water can flow (Dexter et al. 2008; Alaoui et al. 2011). Defining this pore network involves the concept of structural hierarchy that depends on the spatial arrangement of soil particles into secondary units called aggregates (Dexter et al. 2008). These aggregates may vary extensively in size, shape and composition and are highly dependent on soil texture. Soil porosity is also dependent on soil texture. Textural porosity can be described as the void between the primary particles or intra-aggregate pores (micropores) and matrix porosity is the pore space between micro-aggregates and aggregates (macropores) (Alaoui et al. 2011).

Soil macropores have a significant influence on water flow within the soil profile, especially in well aggregated soil. Arvidsson (1997) reported a minor reduction in soil ρ_b may cause a major decrease in soil K_s of several orders. Finer pores are also associated with an increase in contact points between soil particles, which leads to greater internal aggregate strength and lower wet-ability (Ferrero et al. 2007; Goebel et al. 2004). Misra and Sivongxay (2009) applied tap water to a well aggregated soil re-packed at different soil ρ_b (1, 1.1, 1.15 and 1.2 g cm^{-3}) and found that the K_s values increased with decreasing soil bulk density. Thus soil pore networks, particularly macroscopic networks, tend to have important control over soil K_s.

Colloids released from soil aggregates during infiltration using poor quality water, or induced by mechanical disturbance such as rain drop impact, can influence water infiltration due to colloid mobilization and subsequent accumulation (Kjaergaard, 2004). Generally, colloids released within the soil result from degradation of soil aggregates (Oades, 1993). The main mechanisms for aggregate degradation are summarized as: (i) slaking, which is caused by the compression of entrapped air during wetting; (ii) differential swelling of soil particles; (iii) mechanical breakdown of aggregates due to rain drop impact, or cultivation; and (iv) physicochemical dispersion (Alaoui et al. 2011; Rengasamy et al. 1984).

The impact of clay swelling and dispersion on pore blockage is widely documented (McIntyre, 1957; Agassi et al. 1981; Kazman et al. 1983). Yet few studies explain the impact of changes in soil pore networks on the transport of colloidal clays throughout the soil. Rowell et al. (1996) studied the effect of the solution concentration on swelling, dispersion and the movement of clay particles in saline and alkali soils. Their results showed permeability decreased at all concentrations, due to the swelling of clay and subsequent movement of dispersed clay through soil pores leading to soil pores blockage. Mace and Amrhein (2001) used soil K_s as an indicator of pore blockage to examine the impact of dispersed clay on pore plugging during infiltration with saline-sodic water. They found K_s values reduced with increasing sodium adsorption ratio (SAR) as a result of dispersed clay movement and subsequent pore blockage. However, this work has only been conducted on a limited range of soils types and not using the dominant soils exposed to saline-sodic groundwater or coal seam gas associated water in southern Queensland. Thus, this work examines the effect of soil ρ_b of on the transport and subsequent accumulation of dispersed clay within soil cores using a local soil and water containing dispersed clay of various concentrations.

4.2 Materials and methods

4.2.1 Soil and core preparation

Building grade sand obtained from the civil engineering laboratory at the University of southern Queensland (Table 4.1) and a local Red Ferrosol (RF) soil were collected (Section 3.2) and used to create repacked cores to examine the effect of suspended clay concentration on the movement of water and colloids within soil. The RF soil is well structured (Isbell, 2002) and is characteristic of the Toowoomba region (Biggs et al. 2001).

Table 4-1 selected properties of the sand

Properties	*Unit*	*value*
EC (1:5 soil – water ratio)	dS m^{-1}	0.05 ± 0.03
pH (1:5 soil - water ratio)		7.10 ± 0.01
Clay	%	8.15 ± 0.35
Silt	%	27.15 ± 0.55
Sand	%	64.70 ± 0.8
Exchangeable Na concentration	meq 100 g^{-1}	0.05 ± 0.001
Exchangeable Ca concentration	meq 100 g^{-1}	0.20 ± 0.07
Exchangeable Mg concentration	meq 100 g^{-1}	0.50 ± 0.03
Exchangeable K concentration	meq 100 g^{-1}	0.08 ± 0.001
Cation exchange capacity (CEC)	meq 100 g^{-1}	0.99 ± 0.1
Exchangeable sodium percentage (ESP)	%	5.1 ± 0.003

Washed and dried PVC tubes (8 cm high and 5 cm in internal diameter) were used to create repacked soil columns for this experiment. The sand was packed into the columns at soil ρ_b of 1.5 g cm^{-3} and the RF soil was packed into the columns at ρ_b of 1, 1.1, 1.15 and 1.2 g cm^{-3} to provide a range of pore sizes. Soil columns were packed using the method described by Misra and Li (1996) outlined in section 3.3.2. Twelve columns of soil treatment were packed creating a total of 48 columns for the RF soil and 12 columns for the sand.

4.2.2 Suspension preparation

The suspension was prepared using only a single clay type. As this study was investigating the mechanism of soil pore blockage it was assumed that the mineralogy of the suspended clay would not affect blockage. High clay content Black Vertosol (Table 3.1) was air dried and crushed to pass through a 2 mm sieve. A 130 g sample of the sieved soil was then placed in a 500 cm^3 beaker with 400 cm^3 of tap water (EC = 0.7 ± 0.05 dS m^{-1}). The Branson 250 ultrasonic probe (Sonic & Materials, Newtown, USA) was inserted to a depth of 2 cm in the soil-water suspension. Ultrasonic energy was applied to the suspension at a power setting of 80 for 15 minutes. The beaker was then left for 30 minutes on the bench to allow cooling and settling of large particles. The supernatant, containing dispersed clay, was decanted into a 20 L plastic container. This process was repeated using a fresh soil and water suspension until approximately 18 L of suspension was collected. The suspension was then thoroughly mixed and three sub-samples (100 cm^3 each) were taken to measure the suspension concentration by the differences in weight before and after drying the samples at 105 °C for 24 hours. The electrical conductivity (EC) of the undiluted suspension was 1 dS m^{-1} and the pH was 6.95. The bulk concentration of the clay sediments was found to be 90 ± 4 g L^{-1}. The remaining suspension was then diluted using tap water (EC = 0.7 ± 0.05 dS m^{-1}) to create a range of suspension concentrations (5 ± 0.2, 15 ± 3 and 20 ± 2.5 g L^{-1}).

4.2.3 Hydraulic conductivity and sediment load measurement

Good quality water (GQW) free of suspended clay, or clay suspensions (CS) at 3 different concentrations (5, 10 and 20 g L^{-1}) were applied to the soil columns. Three replicates of each treatment were applied. The soil columns were supported by a wooden frame over separate plastic containers of 500 cm^3 capacity and 1500 cm^3 of either GQW or CS after being mixed carefully was applied to the appropriate soil column to undertake the soil K_s measurement. A constant head (1.5 ± 0.3 cm at the soil surface) was maintained during measurement. The leachate (drainage water) was collected for each core to measure the drainage rate and sediment discharge. The discharge was used to estimate soil K_s following Darcy's law for vertically saturated flow of water. Leachate samples were dried in an oven at 105 °C for 24 hours and weighed with an electronic balance (± 0.001g) to determine the sediment discharge.

4.2.4 Soil-water retention

Undisturbed soil samples were taken from the top surface and bottom surface of ρ_b = 1 and 1.2 g cm^{-3} RF soil columns after treatment with GQW and SC = 5 and 20 g L^{-1}. The samples were used to determine if the water quality affected the distribution of specific pore sizes due to pore blockage during hydraulic conductivity measurements. Stainless steel rings (3 cm high and 3 cm inside diameter) with an inside mark at 2 cm were carefully pushed into the top or the bottom of the selected soil column up to the 2-cm mark (Figure 4.1). The steel ring and soil sample was then gently removed by cutting with a saw along the edge of the ring. Soil retained within the ring was then supported by a piece of cheesecloth attached to the outside of the metal tube to prevent any soil loss during subsequent measurement. Two replicate samples were obtained from each end of the original soil column and used for the measurement of soil-water retention using the method suggested by Loveday (1974). Soil within the rings was placed on a porous plate and saturated with tap water via capillary rise for 24 h. The soil and the ring were weighed with an electronic balance (± 0.001 g). The samples were then drained sequentially at each of -2, -4, -6, -8 and -10 kPa water potentials over 72 h by adjusting the height of the porous ceramic plate to create a gravitational head difference from a free water surface at 20, 40, 60, 80 and 100 cm. Samples were weighed after drainage at each water potential to calculate water content. This range of water potential water was chosen to investigate the changes in soil pore classified within the range of soil macropores. Soil-water content was measured gravimetrically after drainage at the lowest water potential (-10 kPa) by removal of the soil from the ring and drying in an oven at 105 °C for 24 h.

4.2.5 Calculation of water-filled pore size

The Washburn (1921) equation was used to calculate the water-filled pore size from the soil-water retention data:

$$P = -\frac{2\gamma\, cos\theta}{r} \tag{4.1}$$

where P = absolute pressure being applied (N m^{-2}), γ = surface tension of water (72 N m^{-1}), θ = contact angle between water and the pore wall (θ = 0) and r = pore diameter (m).

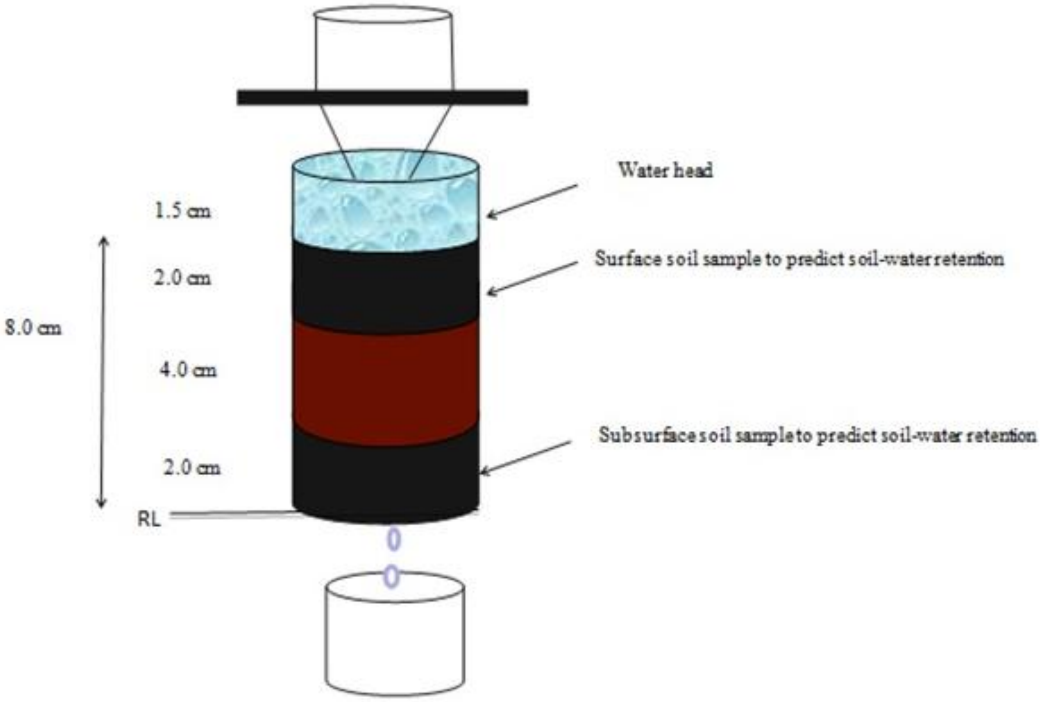


Figure 4-1 Schematic diagram of the soil column showing the position of soil sampling for soil-water retention measurement

The pressure applied at saturation was assumed to be 0.1 kPa. The changes in pore size for each water potential were plotted with the fraction of pore space filled with water, where the volumetric water content at saturation was considered as the total porosity of the soil.

4.2.6 Statistical analyses

Data obtained from this work was analysed using SPSS 16.0. Significant differences between treatments were identified using Univariate ANOVA and the least significant difference method (LSD) at P = 0.05.

4.3 Results

4.3.1 Hydraulic conductivity of soil

The K_s of the sand (315 mm h^{-1}) did not vary significantly with GQW application (Figure 4.2). However, both initial and final K_s were significantly reduced with increasing suspended clay concentrations. For example, final K_s after 8 pore volumes had drained was reduced to 240 mm h^{-1} when the 5 g L^{-1} suspension concentration was applied, but reduced to less than 5 mm h^{-1} by PV10 when the 20 g L^{-1} suspension was used.

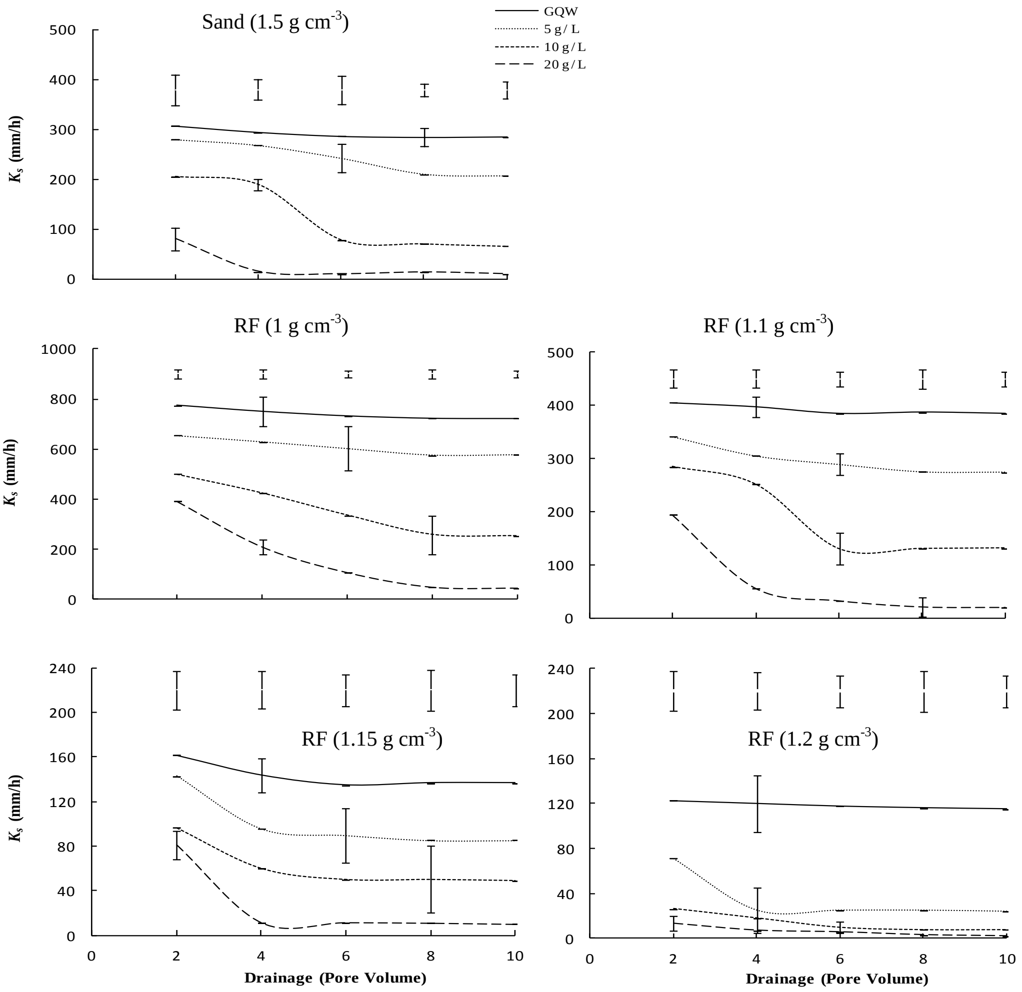


Figure 4-2 Effect of sediment concentration and volume applied on Ks for a sand packed at 1.5 g cm-3 and the RF soil packed at the different bulk densities (1, 1.1, 1.15 and 1.2 g cm-3). Bars placed on the graph are LSD (α= 0.05) between treatments, Bars plotted on curves are LSD between drainage volumes.

Similar results were observed for the RF soil. The K_s decreased significantly with drainage volume, increasing suspension concentration and soil ρ_b (Figure 4.2). When GQW water was used, the K_s did not vary significantly with drainage and for the ρ_b = 1 g cm^{-1} was around 783 mm h^{-1}. However, increasing soil ρ_b to 1.2 g cm^{-3} caused a 86% decrease in the final soil K_s (10 pore volume drained) using GQW water. Applying dispersed clay significantly reduced soil K_s further with the reduction being more severe as the applied suspension concentration increased. A final K_s of 600 mm h^{-1} was observed for the ρ_b = 1 g cm^{-3} treatment when 5 g L^{-1} suspension was applied. However, after the equivalent drainage, the K_s was 13 mm h^{-1} when the 20 g L^{-1} suspension was applied. When the ρ_b was increased to 1.2 g cm^{-3} and 5 g L^{-1} suspension applied, the K_s value reached a steady state of 25 mm h^{-1} after 8 pore volumes had drained. This value was significantly lower at 2 mm h^{-1} after 4 pore volumes of drainage when the 20 g L^{-1} suspension was applied.

It was noted that as the concentration of suspended clay and compaction level increased the steady state K_s of the soil was reached after less drainage had occurred. For instance, when 20 g L^{-1} suspension was applied to the ρ_b = 1 and 1.1 g cm^{-3} treatments, the steady state K_s (5 and 20 mm h^{-1}, respectively) was approached after 8 and 6 pore volumes drainage (Figure. 4.2). However, the steady state K_s (11 and 3 mm h^{-1} respectively) was observed after 4 pore volumes of drainage when the same clay suspension concentration was applied to ρ_b = 1.15 and 1.2 g cm^{-3} treatments.

4.3.2 Sediment retained in soil columns

The proportion of applied sediment retained in the sand and the RF soil during the K_s measurement is shown in Figure. 4.3. No sediment was applied in the GQW treatments and hence these treatments are not included in the graphs.

For the sand, increasing sediment concentration in the applied solution significantly increased the percentage of sediment retained in the columns. When the 5 g L^{-1} suspension was applied to the sand, only 5% of the applied clay sediment was retained in the columns after 2 pore volumes had drained but 60% was retained after 8 pore volumes. When the 20 g L^{-1} suspension was applied, 60% of the applied sediment was retained after 2 pore volumes had drained but this increased to 95% after 8 pore volumes. Hence, as suspended clay concentration in the applied solution increased the sediment retained in the soil reached a maximum after less drainage.

For the RF soil, sediment retained in columns increased significantly as the suspended clay concentration in the applied solution and soil ρ_b increased. When the 5 g L^{-1} suspension was applied to the RF soil packed at 1 g cm^{-3}, little sediment was retained until at least 4 pore volumes had drained after which approximately 11% was retained. When 20 g L^{-1} suspension was applied the sediment retained was higher at 45% after 4 pore volumes were drained. Sediment retained increased as soil ρ_b increased. When 5 g L^{-1} suspension was applied to compacted soil (1.2 g cm^{-3}), sediment retained in the columns was 42% after 2 pore volumes of drainage and increased to 83% after 4 pore volumes. This compacted soil retained all applied sediment when 20 g L^{-1} was applied and more than 2 pore volumes of drainage had occurred.

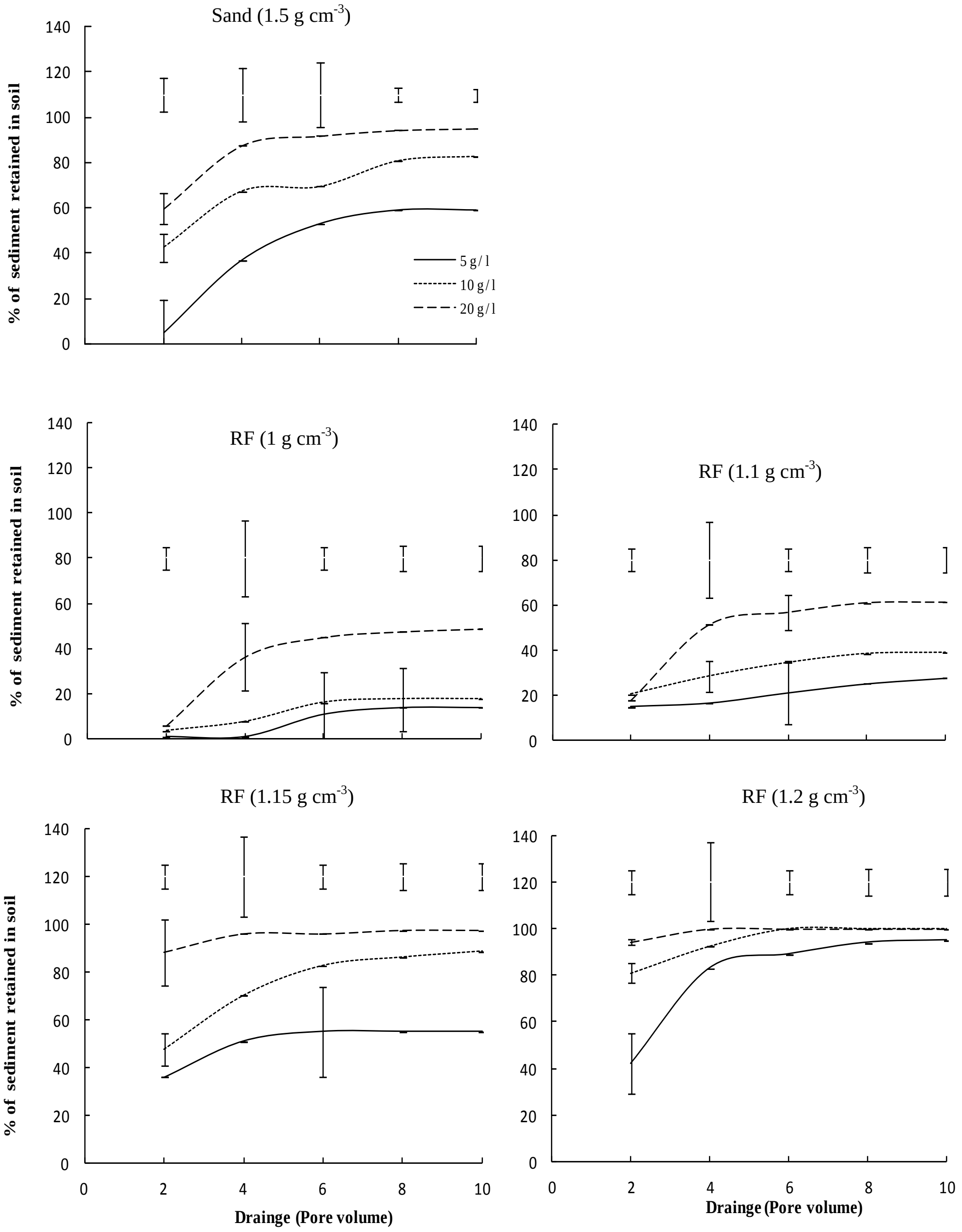


Figure 4-3 Effect of sediment concentration and applied water on sediment retention in sand (ρb = 1.5 g cm-3) and Red Ferrosol (ρb = 1, 1.1, 1.15 and 1.2 g cm-3). Bars placed on the graph are LSD (α = 0.05) between treatments and bars plotted on curve are LSD between drainage volumes.

4.3.3 Soil-water retention

Soil water retention functions (Figure 4.4) were significantly affected by suspended clay concentration of the applied solution, soil ρ_b and soil location of sampling within the soil columns. There was no significant difference in the soil-water retention functions of the surface and subsurface layers when GQW was applied (Figure 4.4). However, the surface layer in compacted columns retained more water content compared to the subsurface layer when sediment was applied. In the lower ρ_b

treatments, less water was held in the surface than the subsurface samples. The surface soil from columns compacted at ρ_b = 1 g cm^{-3} and treated with the 20 g L^{-1} suspension retained 50% more water at 2 kPa matric suction compared to the columns treated with GQW. At 10 kPa, water held in the surface layer of the column treated with 20 g L^{-1} suspension was twice that held in the column where GQW was applied.

As the ρ_b increased, the changes in retained water were also significantly affected by suspended clay concentration of the applied solution. The water retained at 2 kPa in the surface layer of the column packed at 1.2 g cm^{-3} and treated with solution of 20 g L^{-1} was 53% more than that of the column treated with GQW. This was consistent at all other higher matric suctions. A similar scenario was found for the subsurface soil samples. For instance, at 10 kPa the surface soil compacted at 1 g cm^{-3} and treated with the 20 g L^{-1} suspension held 160% more water than column treated with GQW (Figure 4.4). Increasing soil bulk density also had significantly increased water retained in all treatments. For example, surface samples from columns packed at 1.2 g cm^{-3} and treated with GQW held 60% more water at 10 kPa than those columns packed at 1 g cm^{-3}.

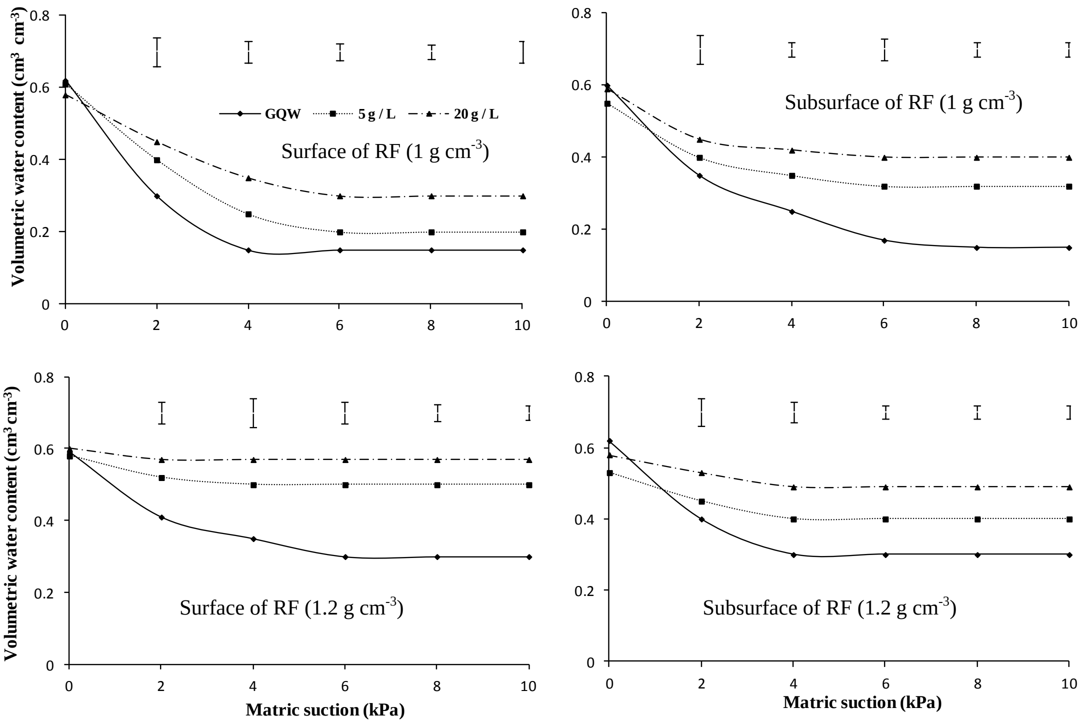


Figure 4-4 The effect of sediment concentration, packing density and depth within soil column on soil-water retention functions for RF soil. Bars plotted on curve are LSD matric suctions.

4.3.4 Pore size distribution

The size distribution of drained pores (Figure 4.5) was derived from the soil-water functions (section 4.3.3). Increasing soil ρ_b and the concentration of the suspended clay applied resulted in a significant decrease in the larger pore sizes. This effect occurred within samples for compacted soil but in subsurface samples for lower ρ_b soil (Figure 4.5). For instance, the fraction of drained pores that have diameter ≥ 144 µm was 0.8 for the surface sample from the column at ρ_b = 1 g cm^{-3} and treated with GQW but was 0.48 for the same treatment when the 20 g L^{-1} suspension was applied.

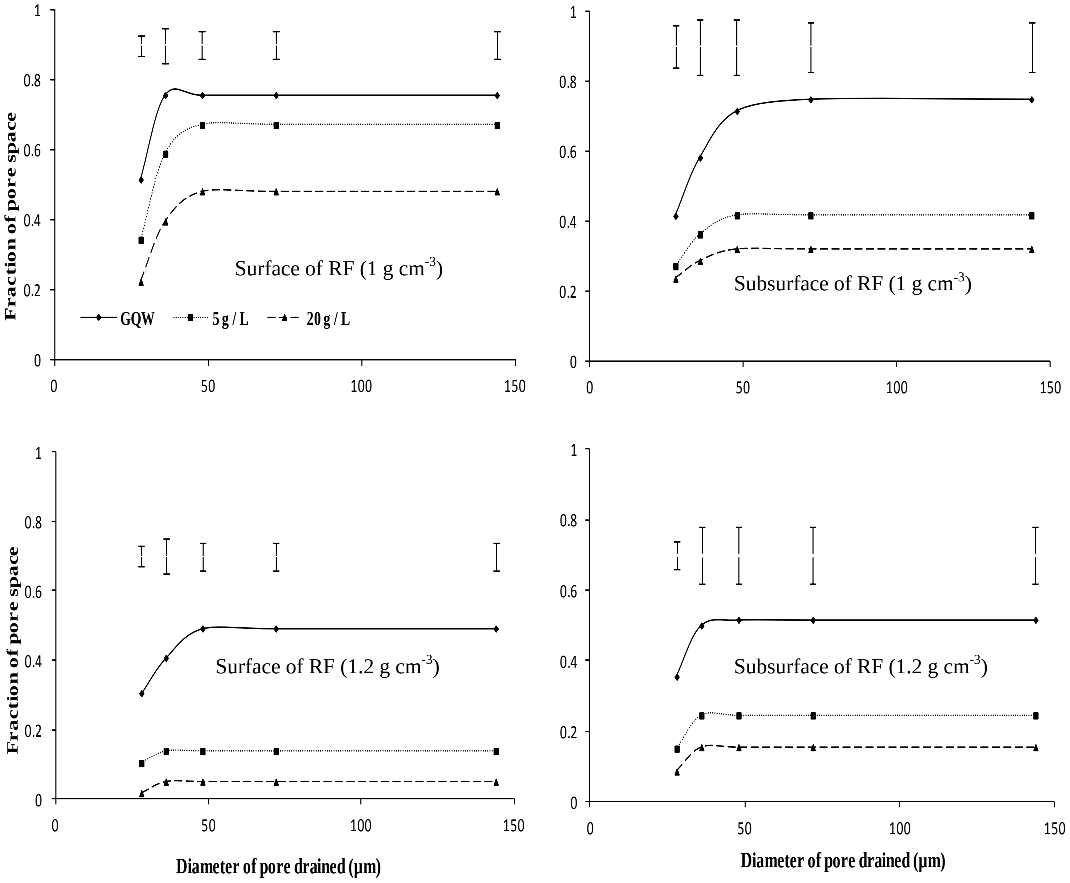


Figure 4-5 Fraction of pores drained of surface and subsurface samples for RF soil packed at 1 or 1.2 g cm-3 and treated with QGW, 5 or 20 g L-1 clay suspension. Bars placed on the graph are LSD (α = 0.05) between treatments.

For the soil column compacted at ρ_b = 1.2 g cm^{-3}, the fraction of pores ≥ 144 µm decreased significantly from 0.45 when GQW was applied to 0.05 when the 20 g L^{-1} suspension was applied. However, for subsurface soil packed at 1 g cm^{-3}, the fraction of pore size ≥ 144 µm was 0.78 when treated with GQW but decreased to 0.32 when 20 g L^{-1} suspension was applied. There was also a significant difference between the fraction of pore size ≥ 144 µm for the subsurface soil packed at 1.2 g cm^{-3} when treated with GQW (0.5) and the 20 g L^{-1} suspension (0.15).

4.4 Discussion

4.4.1 Effect of soil bulk density on retained sediment and saturated hydraulic conductivity

In general, the results showed a decrease in sediment concentration of the drainage water during the leaching process, particularly for compacted soil with high suspended clay concentration solution applied. The sand has a stable structure and infiltration when GQW was applied. However, when sediment was applied, the reduction in outflow clay sediments was directly related to the accumulation of the clay sediment within flow paths. This led to a reduction in the flow path, or pore network, through pore clogging (Rodgers et al. 2006).

Approximately half (45%) of the sediment was retained in the RF soil with ρ_b = 1 g cm^{-3} after 10 pore volumes had drained after the highest (20 g L^{-1}) sediment concentration was applied. That is a consequence of soil with low ρ_b containing a larger proportion of large pore sizes, as demonstrated by the fraction of pores 50-144 µm being 0.8. This has likely allowed easy passage of colloids throughout the macroscopic soil pore network. However, the fraction of pores between 50-144 µm significantly decreased for surface and subsurface soils to 0.48 and 0.32 after the application of 10 pore volumes drained of 20 g L^{-1} suspension solution. This suggests that clay sediments were transported past the surface further into the soil profile by larger pores. Pilgrim and Huff (1982) suggested that clay sediments during runoff processes in the field could accumulate within soil profiles in certain layers depending on soil pore size and the size of migrated clay sediments. However, Arshad et al (1999) suggested that water movement within the soil profile is controlled by the structural aspect of the most limiting soil layer.

The accumulation of sediment within the of soil columns and reductions in the fraction of large soil pores particularly in subsurface layers resulted in significant reductions in K_s from 783 to 48 mm h^{-1} for density 1 g cm^{-3}. However, increasing the ρ_b decreases the proportion of large pores, while developing a higher proportion of small pores, as compaction of aggregated soil increases the contact surface area between soil aggregates leading to reduction in pore space between soil aggregates (Van Asch, 2002 and Gupta et al. 1989). Therefore, the soil compacted at ρ_b = 1.2 g cm^{-3} is expected to contain a larger proportion of small pores which could be easily blocked by the suspended clay during leaching processes. In this work the proportion of large soil pores with (diameter 50-144 µm) was 50% smaller in compacted soil (both surface and subsurface layers) compared to the lower ρ_b soil when GQW was used (no sediment) suggesting this reduction was produced by increasing soil ρ_b. Consequently, the K_s reduced by 80% in the compacted soil column compared to that packed at 1 g cm^{-3}, indicating that a small increase in soil ρ_b results in a larger reduction in the proportion of macropores. Furthermore, that K_s reduction was mainly related to changes in the proportion of macropores as GQW does not contain clay.

After 10 pore volumes of 20 g L^{-1} suspension had drained through the soil columns packed at 1.2 g cm^{-3}, the proportion of large pores (50-144 µm) declined from 0.5 to 0.05 and 0.15 for surface and subsurface layers respectively, indicating severe pore clogging, particularly at the surface. McGechan and Lewis (2002) explained the main type of physical capture mechanisms controlling colloid movement in soil and other porous media as straining, where the size of soil pores is less than the size of percolated particles so that the particles are not allowed to pass and are captured within surface soil pores. Hence, even though a significant reduction in the fraction macroporosity occurred in the subsurface sample, the surface soil will ultimately behave as the throttle to infiltration. For all soils and soil ρ_b treatments the concentration of sediment in the drainage water declined over time indicating some suspended materials were arrested within the soil pores..

This preliminary work has shown that soil pore size has a significant effect on sediment clay migration and accumulation within soil profiles of the RF soil. However, deriving soil pore size distribution from soil-water retention does not provide sufficient data to explain the mechanisms associated soil pore blockage. In addition, clay sediment migration and soil pore blockage is a common process in saline-sodic soils where every soil behaves differently depending on clay mineralogy and the quality of irrigation water applied. Therefore, these issues will be further investigated in chapter 5 and 6 by undertaking a resin impregnation method that provides more details about the mechanisms of soil pore blockage.

4.4.2 Saturated hydraulic conductivity as a function of bulk density and sediment concentration in applied suspension

There was a significant interaction between soil ρ_b and applied clay sediment concentration on K_s. For example, the smallest K_s (2 mm h^{-1}) was obtained when soil compacted at 1.2 g cm^{-3} was treated with the 20 g L^{-1} suspension. This K_s value was significantly lower than when the 20 g L^{-1} suspension was applied to soil columns packed at 1, 1.1 or 1.15 g cm^{-3}. A strong linear negative correlation (R^2 = - 0.737) was found between the soil K_s and soil ρ_b while a weaker linear correlation (R^2 = - 0.487) was obtained between soil K_s and sediment concentration in the applied solution. This indicates the significant effect of soil ρ_b on water and suspended clay movement in soil.

Table Curve 3 D v4.001 software was used to investigate the influence of ρ_b and sediment concentration in the infiltrating solution on soil K_s (Figure 4-6). The simplest equation predicted by the model was of the form:

$$K_s = e^{(a+c\rho b+dy^3)} \qquad 4.2$$

Where: K_s is the saturated hydraulic conductivity (mm h^{-1}), ρ_b is soil bulk density (g cm^{-3}), y is sediment concentration (g L^{-1}), a, b and c are constants. The values of produced by the model were a = 10.58, b = 0.0714, c = 3.255 and coefficient of determination (R^2) = 0.9252.

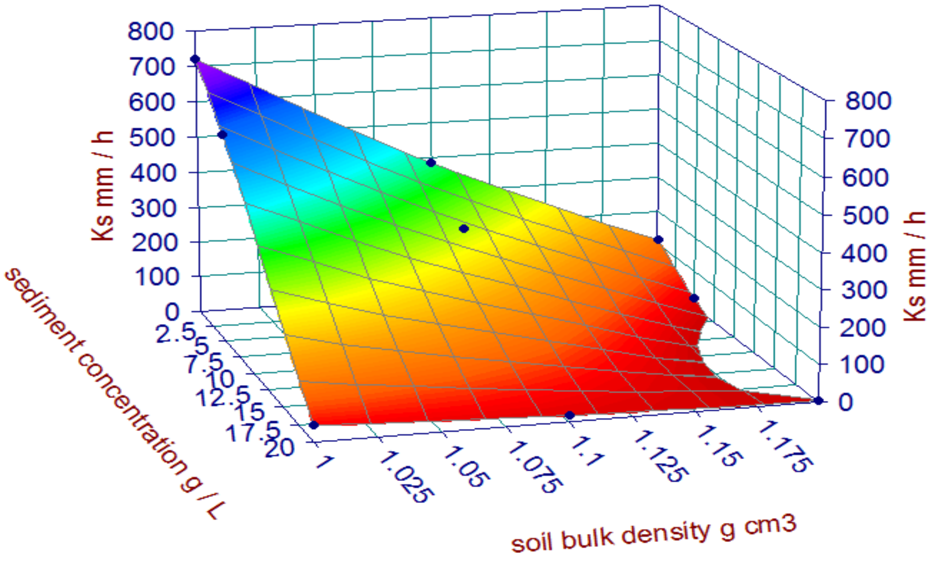


Figure 4-6 Changes in Ks as a function of sediment concentration of applied solution (GQW, 5 and 20 g L-1) and soil bulk densities (1, 1.1 and 1.2 g cm-3).

4.5 Conclusion

This experiment has shown that soil pore size and the sediment concentration of the applied suspension has a significant impact on dispersed clay movement, pore blockage and K_s values within the soil columns. When the soil was packed at low ρ_b (1.0 and 1.1 g cm^{-3}) the large pores allowed clay sediment to pass through the soil columns and drain in the leachate. By increasing the soil ρ_b to 1.15 and 1.2 g cm^{-3} the soil retained a higher percentage of the applied sediment, even at low clay sediment concentration solutions, indicating clay sediment accumulation within soil pores and suggesting that increasing soil ρ_b increased the proportion of small pores that were easily blocked.

Increasing suspended clay concentration applied and soil ρ_b led to a decrease in soil K_s and affected soil-water retention and pore size distribution. The soil pore blockage was more significant near the soil surface for compacted soil and in the subsurface for soil packed at low bulk density. This suggested that clay sediments were transported past the surface further into the soil profile by larger pores at low ρ_b. However, for the compacted columns, the surface soil behaves as a throttle to sediment movement and infiltration.

Chapter 5: Understanding the Relationship Between the Rate of Ion Exchange and Changes in Saturated Hydraulic Conductivity

5.1 Introduction

Sodicity and salinity are the main water quality concerns in irrigated soils receiving lower quaility water (Oster and Schroer, 1979; Ayers and Tanji, 1999). For example, Mantell et al. (1985) found significant increases in soil sodicity in soil irrigated with saline-sodic water (EC = 3-8.5 dS m^{-1} and SAR = 14-26). Sharma and Manchanda (1996) indicated that using sodic water leads to reductions in soil hydraulic conductivity (K_s), reductions in air and water movement within the soil and poor plant growth. However, the electrical conductivity (EC) of the applied water is also a major factor affecting soil structure (section 2.6), whereby sufficient EC in the soil solution can maintain soil structure in the presence of excess exchangeable sodium on clay exchange sites. The EC required to maintain stable infiltration in the presence of sodium is known as the threshold electrolyte concentration (TEC) (Quirk and Schofield; Ezlit et al. 2013). There is no clear TEC boundary because structural degradation occurs gradually as sodium increases in the soil so an arbitrary 15-25% reduction in hydraulic conductivity is commonly chosen to define the TEC (section 2.7). For example, Bennett and Raine (2012) used a 20% reduction in K_s to define the TEC function. They also found that soils with similar clay properties and in the same soil classification order may have different TEC curves, suggesting that other soil properties also affect the K_s and hence, TEC.

Ion exchange in soil is a reversible procedure where cations are exchanged between liquid and solid phases (Sposito, 1989). According to diffuse double layer (DDL) theory, the charge of the cations affects the size of the DDL and final structural stability of the soil. To satisfy the cation exchange capacity (CEC), double the amount of monovalent cations (Na^+ and K^+) are required on the exchange surface to neutralise the negative charge, as compared to divalent cations. Therefore a Na-dominated DDL will be wider than a Ca-dominated DDL and this makes the clay swell and more likely to disperse. Increasing the osmotic pressure, via increasing soil solution EC, the Na-dominant DDL can be compressed (Quirk and Schofield 1955). Therefore, soil permeability depends greatly on the type of ions present in irrigation water and their concentration (Quirk and Schofield 1955; McNeal and Coleman 1966 and Ezlit et al. 2013).

Irrigation waters with high sodium adsorption ratio (SAR) increase the risk of clay swelling and dispersion as high sodium water leads to an increase in soil exchangeable sodium percentage (ESP). The United States Salinity Laboratory Staff (1954) suggested an ESP of 15 % was the critical level above which soil structure could be damaged. However, the presence of a threshold and / or the previous threshold value is not universally agreed (Summer, 1993). The ESP border between unstable and stable conditions differs from one soil to another depending on clay content and type (Pratt and Suarez, 1990), the net negative charge of the clay present (Marchuk and Rengasamy 2012) and the irrigation water quality being applied (Sumner 1993).

In an early study (Lai and Jurinak 1972) dealing with the movement and interaction of Na^+ and Ca^{2+} it was assumed that equilibrium was achieved by exchange between the two cations on the soil surface and in the soil solution. Shackelford (1999) suggested that EC breakthrough curves could be used as an indicator of chemical equilibrium between the effluent and influent solutions for soil columns. However, Wieck (2012) showed that the chemical equilibrium between the applied and drained solutions was not approached even when the EC of the effluent was normalized with the EC of the influent. This suggests that the soil hydraulic system is likely operating on multiple scales, potentially with macroscopic flow between aggregates and microscopic flow within aggregates (Wieck 2012; Bullock and Thomason 1979; Van Genuchten 1976). Thus, it could be expected that the soil solid-solution phase chemical equilibrium state is also likely to operate on multiple scales. Of importance to soil TEC then is the requirement of soil macroscopic and microscopic aggregate chemical equilibrium for steady state hydraulic conductivity of the soil pore network when saline-sodic water is applied. Therefore, this study examines the importance of ion exchange of the soil pore network on steady state soil K_s using saline-sodic water.

5.2 Material and methods

Three surface (0 - 100 mm) clay soils each with different CEC were used to examine the effect of saline-sodic irrigation water and soil bulk density on soil chemical equilibrium and K_s steady state. These soils were classified as a Red Ferrosol (RF), Grey Vertosol (GV) and Black Vertosol (BV). The soils were collected and prepared as explained in section 3.2. Selected properties of the soils are given in Table 3.1. The soil columns were prepared as per section 3.3.1. Columns of each soil were packed at two bulk densities (ρ_b = 1 and 1.2 g cm^{-3}). Four replicates were conducted producing 32 columns per soil and a total of 96 columns for the three soils.

5.2.1 Soil pore size measurement

One extra replicate of each soil and bulk density column was dried with acetone and impregnated under vacuum with a mixture of polyester resin (40%), catalyst (50%), hardener (5%) and green-yellow florescent dye (5%) following the Murphy (1986) method described in section 3.3.7. The top surface of each impregnated soil column was smoothed and polished using sand paper. Surfaces of sections were viewed under a microscope set to a magnification of 65 X and a scale was burned into the image. Images were analysed using TBitmap software (NCEA 2013) to determine the percentages of soil macropores (> 30 µm), mesopores (0.2 – 30 µm) and micropores (< 0.32 µm) using a classification procedure developed by Kay (1997).

5.2.2 Saturated hydraulic conductivity, leaching and exchange cations

Four soil columns for each treatment were slowly wetted from the bottom and then saturated for 24 hours in the corresponding solutions intended for measuring soil K_s. The soil columns were then supported with a wooden frame placed above 300 cm^3 plastic containers. A constant water head (1.7 ± 0.4 cm) was then maintained over the columns. Up to 10 pore volumes of the water treatment was applied to each soil columns. The pore volume for the soil columns was 97.6 and 85.8 cm^3, respectively for ρ_b = 1 and 1.2 g cm^{-3} treatments. The effluent leachate was measured and separated at each pore volume and Darcy's Law for vertical saturated flow of water was used to calculate soil K_s. The EC and concentration of Na^+, Ca^{2+}, Mg^{2+} and K^+ in the leachate was determined for each pore volume using the procedure reported in section 3.3.6

Two replicates from each treatment column were left for three days after the final soil K_s measurement to drain the free water. The cheesecloth was removed from the columns and the soil was taken out of the PVC tubes. Sub-samples were taken from the soil surface (0–2 cm), middle (4–5 cm) and from the bottom (7- 8 cm) of each columns. The sub-samples were used for measuring exchangeable cations (Na^+, Ca^{2+}, Mg^{2+} and K^+) using the method described in section 3.3.6.

5.2.3 Statistical analyses

A completely randomized design was used in this experiment. Data are expressed as the mean of five replicates of each parameter studied. Statistical analyses were conducted using SPSS v19 for windows 7 (Cramer, 2004). One way ANOVA was used to study the effect of water quality through each pore volume. Two-way ANOVA was used to study the effect of both water quality and soil bulk density on each parameter. Also Two-way ANOVA was also used to study the effect of both soil depth and soil bulk density on exchangeable cations. Least significant difference (LSD) was used to compare the means with a probability level of 5%.

5.3 Results

5.3.1 Pore size distribution

Image analysis of the soil columns surface (Figure 5.1) prior to water application indicated that the low ρ_b (1 g cm^{-3}) soils had a higher percentage (47-53%) of soil macropores (> 30 µm) compared to the higher ρ_b (1.2 g cm^{-3}) soils (21-28%). The compacted soils had a higher percentage of soil micropores (Table 5-1).

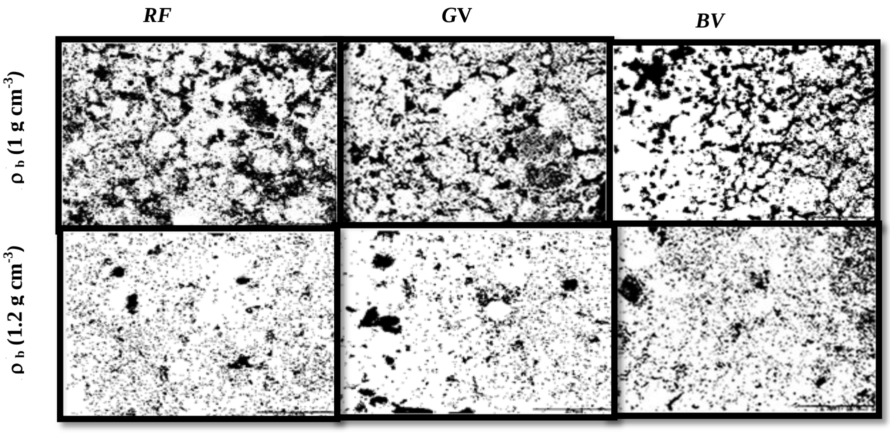


Figure 5-1 Microscope images of horizontal cross-sectioned for RF, GV and BV soils packed at 1 and 1.2 g cm-3.

Table 5-1 Pore size distributions of RF, GV and BV soil (std deviation in brackets)

	% of soil macropores		% of soil mesopores		% of soil micropores	
Soil ρ_b	1 g cm^{-3}	1.2 g cm^{-3}	1 g cm^{-3}	1.2 g cm^{-3}	1 g cm^{-3}	1.2 g cm^{-3}
RF	53 (±4)	28 (±1)	28 (±1)	30 (±4)	19 (±0.9)	42 (±4)
GV	48 (±5)	23 (±0.8)	32 (±0.5)	35 (±2.5)	20 (±1)	42 (±5)
BV	47 (±3.5)	21 (±2)	32 (±3)	32 (±0.9)	21 (±2)	47 (±3)

5.3.2 Electrical conductivity and saturated hydraulic conductivity of soils

The general trend observed for all soils used in this trial was that effluent EC was high for the first pore volume (PV) of drainage then decreased over the next 1-3 PV of drainage to EC values similar to the applied solutions (Figure. 5-2). However, for each soil the peak EC of the drainage water was significantly ($P<0.05$) higher for the ρ_b = 1 g cm^{-3} columns compared to the ρ_b = 1.2 g cm^{-3} columns. The lower density columns for each soil appeared to equilibrate soil EC with the applied solution after 2PV of drainage had occurred for the RF and GV soils and after 3 PV of drainage for the BV soil. By contrast, the high density columns for each soil required approximately 4 PV of drainage to equilibrate. Hence, the leaching of salts from the soil columns occurred more slowly with increasing soil bulk density producing lower peak EC discharges but with elevated discharges occurring over a larger drainage volume.

Significant differences in K_s were found between the three soils. The K_s typically decreased with increasing soil ρ_b and applied SAR for all soils (Figure 5-3). Steady state K_s when GQW was applied to the ρ_b = 1 g cm^{-3} RF soil was 782 mm h^{-1} compared to 160 mm h^{-1} for the GV and only 70 mm h^{-1} for the BV soil (Figure 5-3). The K_s did not vary significantly with increasing volume of GQW application for any soil or density.

Increasing the SAR of the applied water generally reduced the K_s. However, the magnitude of the effect varied between soils and packing densities. For example, there was no significant difference in K_s of the RF soil when GQW or SAR 10 water was applied. However, applying SAR 10 water to the BV soil significantly reduced both the initial and final K_s compared to the GQW application. In all soils and densities, applying the SAR 50 water significantly reduced the K_s compared to the GQW application.

Increasing the soil density was found to increase the amount of sodic water that was required to be applied before the final K_s was reached. For each soil at ρ_b = 1 g cm^{-3}, the final K_s was reached after 2 PV of drainage. However, at ρ_b = 1.2 g cm^{-3} the final K_s was not reached in the GV and BV soils until 4 and 5 PV had been applied, respectively.

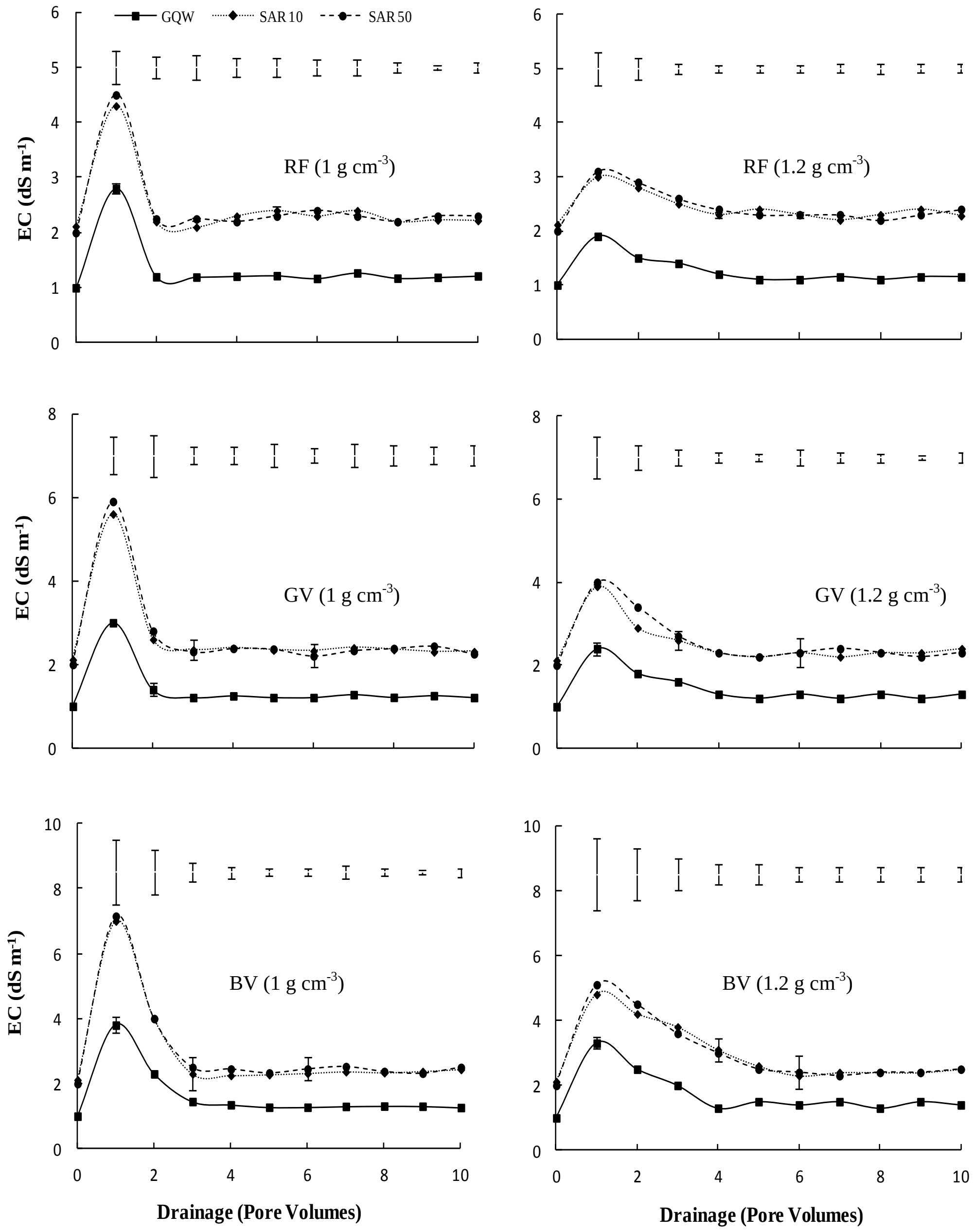


Figure 5-2 Changes in EC with drainage volume for three soils (RF, GV and BV) packed at two bulk densities (1 and 1.2 g cm-3) and treated with different water qualities (GQW, SAR 10 and SAR 50). Bars placed on the graph are LSD ($\alpha = 0.05$) between treatments. Bars placed on the graph are LSD ($\alpha = 0.05$) between treatments. Bars plotted on curve are LSD within treatments.

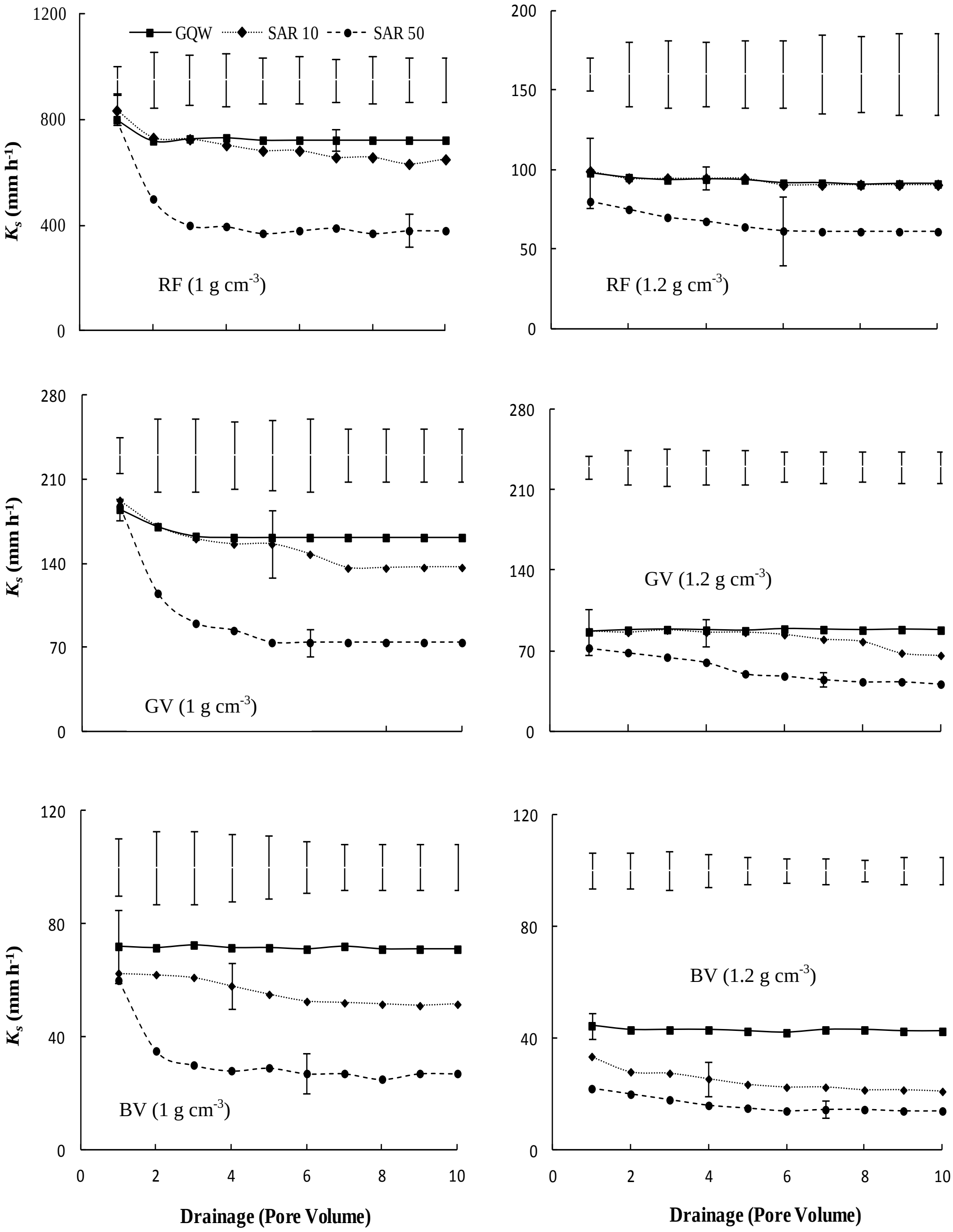


Figure 5-3 Changes in Ks with drainage volume for three soils (RF, GV and BV) packed at two bulk densities (1 and 1.2 g cm-3) and treated with different water qualities (GQW, SAR 10 and SAR 50). Bars placed on the graph are LSD (α= 0.05) between treatments. Bars placed on the graph are LSD (α=0.05) between treatments. Bars plotted on curve are LSD within treatments.

5.3.3 Cation concentration in the leachate

The general trend for cation concentration of the leachate followed that observed for EC (section 5.3.2). Elevated cation concentrations were measured in the first 1-2 PV of drainage for all treatments (Figure 5-4 to 5-7). The initial flushing of ions out of the columns generally required 2 PV of drainage in the low density (ρ_b = 1 g cm^{-3}) columns compared with 3-6 PV in the higher density (ρ_b = 1.2 g cm^{-3}) columns. Increasing the soil density also resulted in slightly higher, and in some cases significant differences in the concentration of Na^+ and Ca^{2+} ions in the leachate, as the drainage volume increased (Figure 5-4 to 5-5). However increasing soil density had no effect on the leaching of Mg^{2+} and K^+ ions for any soil (Figure 5-6 and 5-7). Increasing the EC and SAR of water applied was found to increase the extraction of all cations in each soil. Significant differences were found in the Na^+ concentration after 10 PV between the GQW, SAR 10 and SAR 50 treatments. However, there was generally no significant difference in the Ca^{2+}, Mg^{2+} and K^+ concentrations after 10 PV for the SAR 10 and 50 treatments (Figure 5-5 to 5-7).

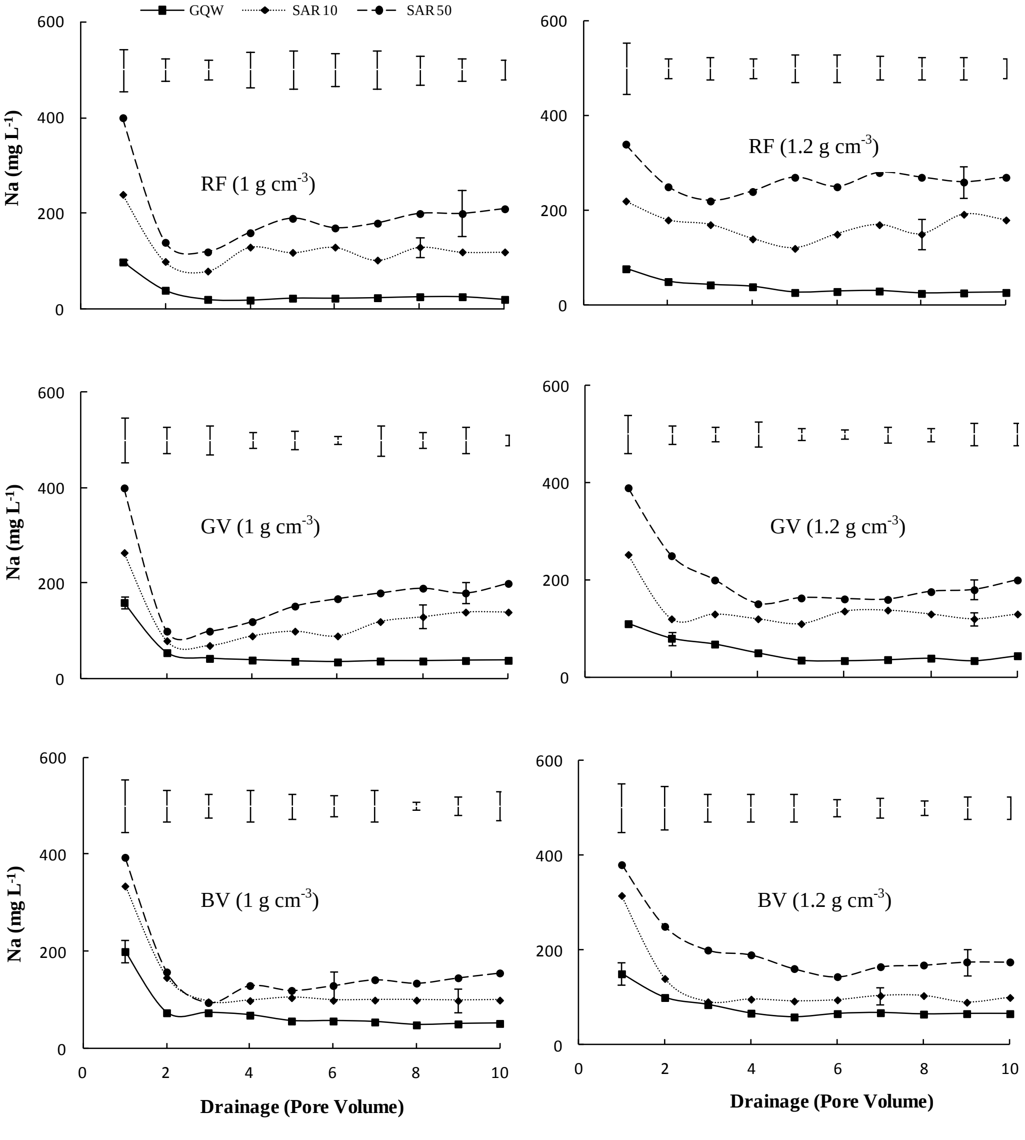


Figure 5-4 Changes in Na+ concentration with drainage volume for the three soils (RF, GV and BV) packed at two bulk densities (1 and 1.2 g cm-3) and treated with different water qualities (GQW, SAR 10 and SAR 50). Bars placed on the graph are LSD (α = 0.05) between treatments. Bars plotted on curve are LSD within treatment.

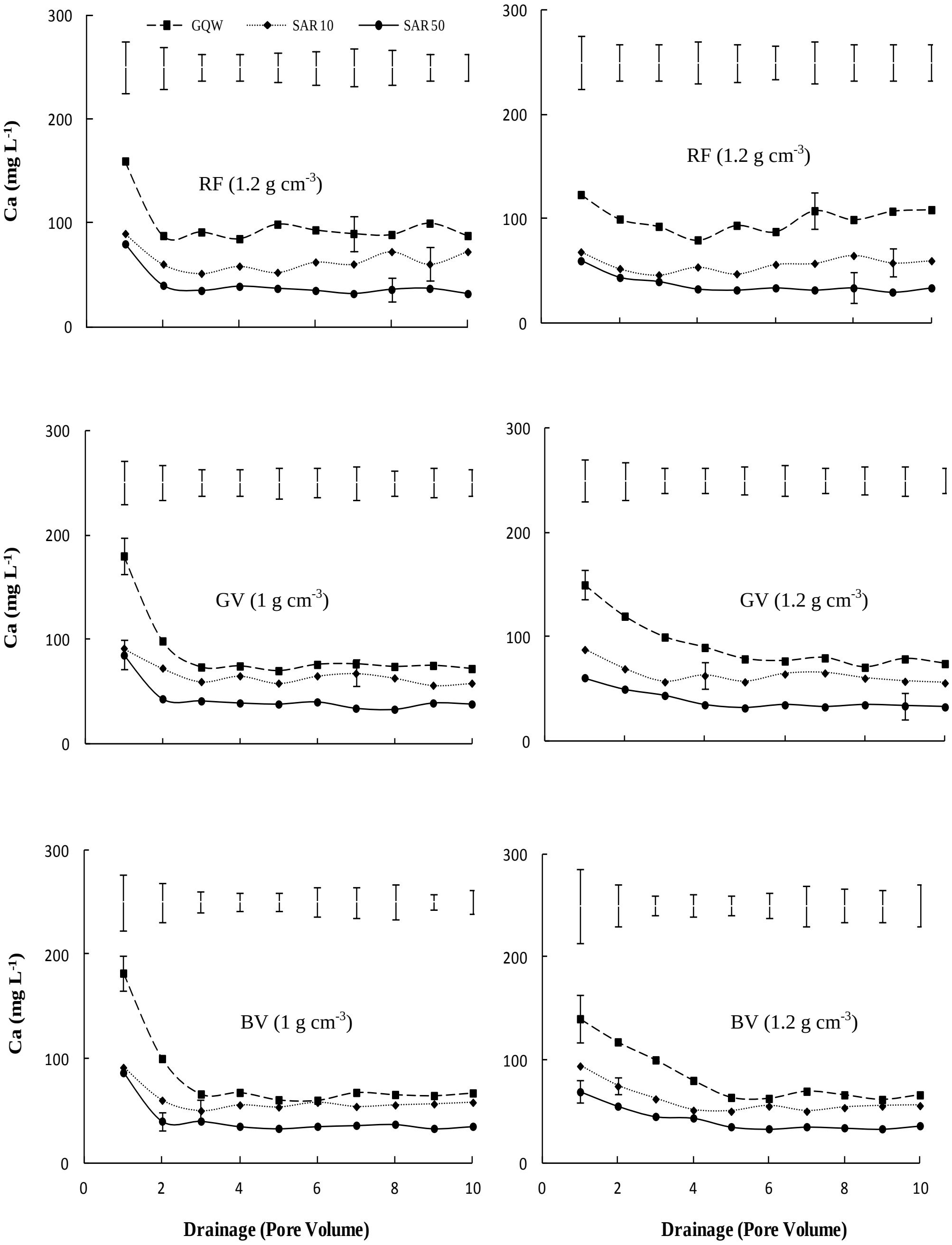


Figure 5-5 Changes in Ca2+ concentration with drainage volume for the three soils (RF, GV and BV) packed at two bulk densities (1 and 1.2 g cm-3) and treated with different water qualities (GQW, SAR 10 and SAR 50). Bars placed on the graph are LSD ($\alpha = 0.05$) between treatments. Bars plotted on curve are LSD between within treatment.

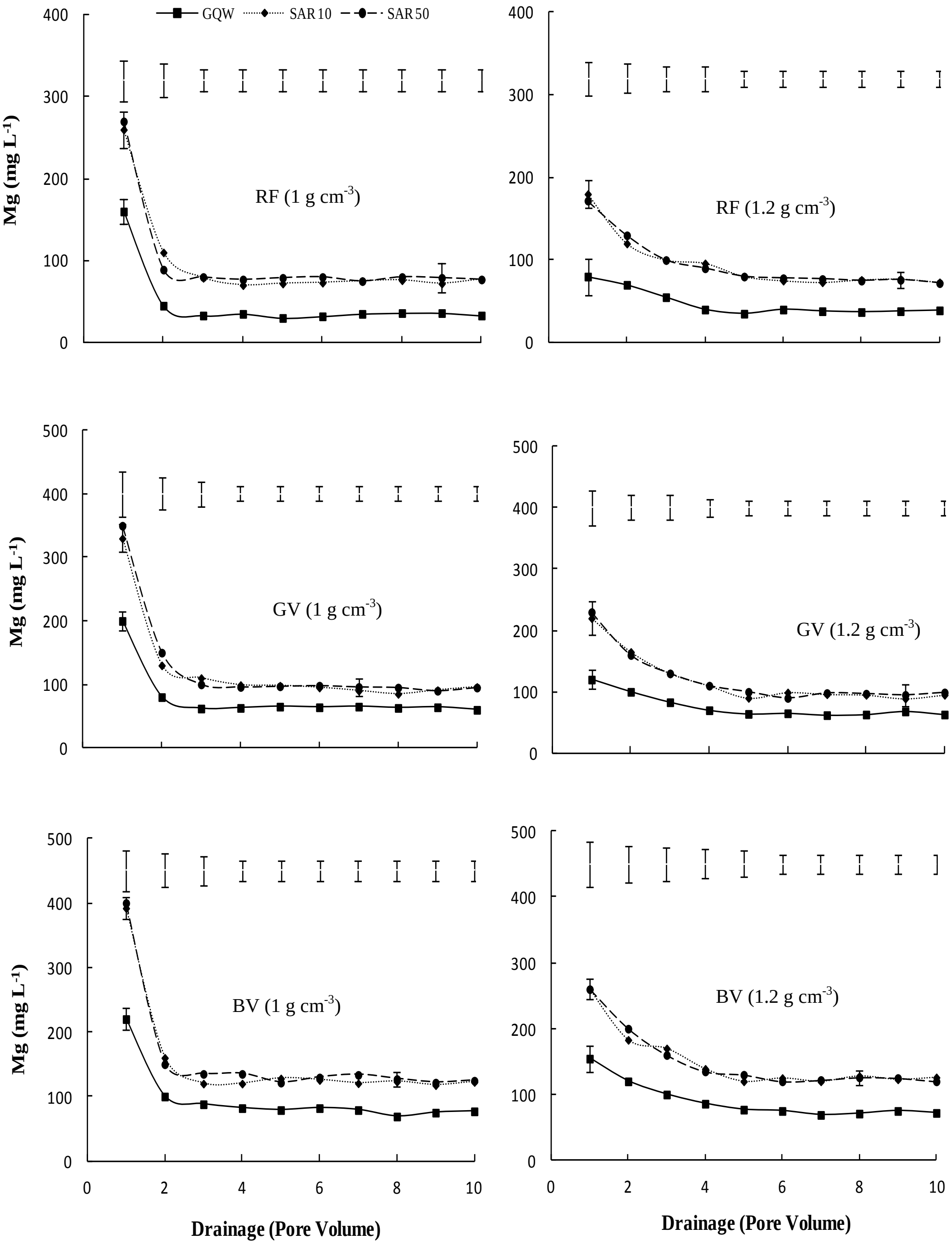


Figure 5-6 Changes in Mg2+ concentration with drainage volume for the three soils (RF, GV and BV) packed at two bulk densities (1 and 1.2 g cm-3) and treated with different water qualities (GQW, SAR 10 and SAR 50). Bars placed on the graph are LSD (α = 0.05) between treatments. Bars plotted on curve are LSD between within treatments.

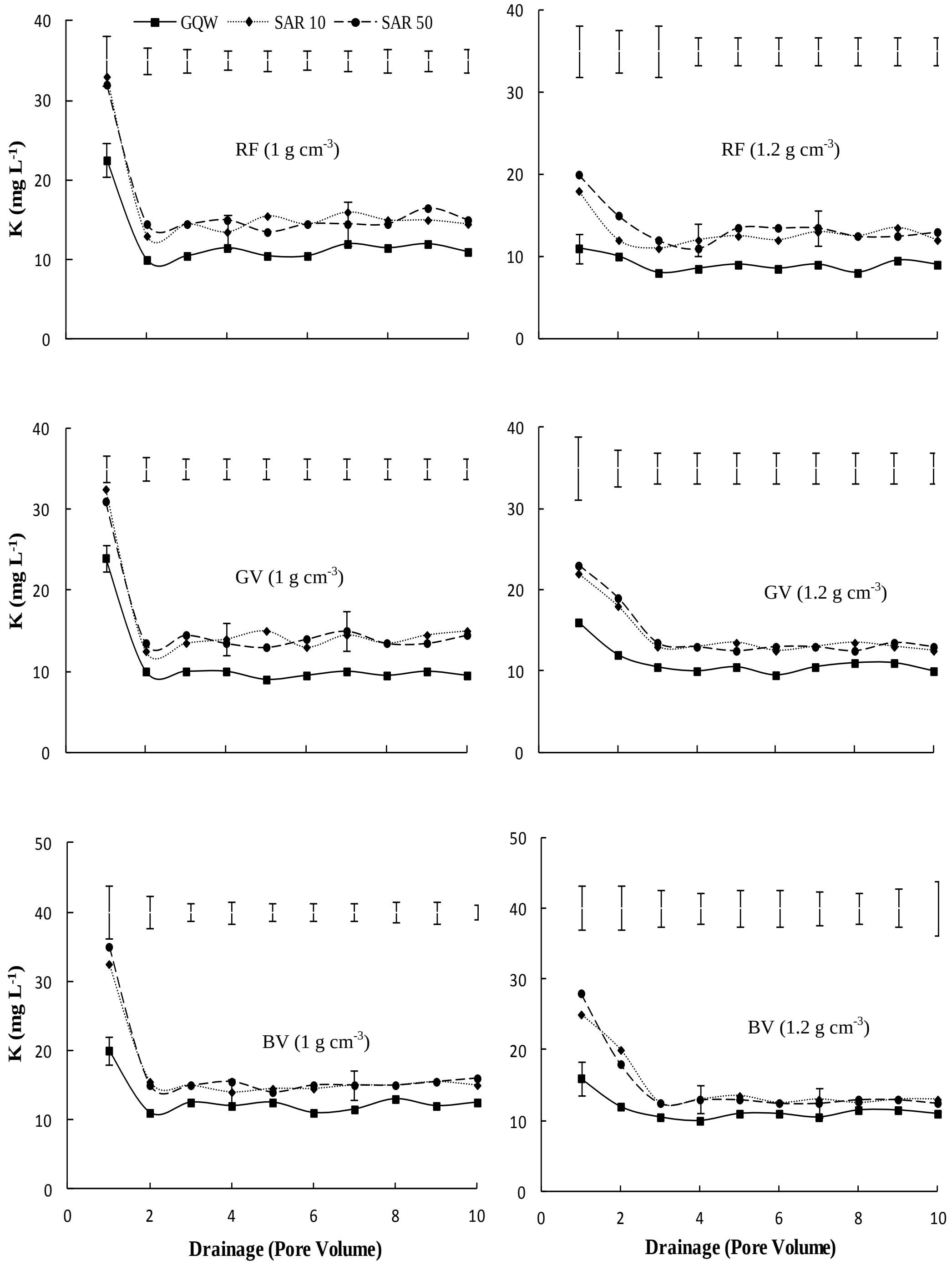


Figure 5-7 Changes in K+ concentration for the three soils (RF, GV and BV) packed at two bulk densities (1 and 1.2 g cm-3) and treated with different water qualities (GQW, SAR 10 and SAR 50). Bars placed on the graph are LSD (α = 0.05) between treatments. Bars plotted on curve are LSD within treatments.

The average cumulative extraction of Mg^{2+} and K^+ ions was calculated for the GQW and SAR 50 treatments (Figure 5.8). Even though Mg^{2+} and K^+ ions were not added to the columns in these treatments, this showed that extraction of these ions increased significantly with addition of the saline-sodic water but was affected by the soil bulk density.

The cumulative Mg^{2+} and K^+ removed in the leachate decreased significantly ($P<0.05$) with increasing soil bulk density when either GQW or SAR 50 water was applied (Figure. 5.8). For example, the total Mg^{2+} removed after 10 PV of drainage from the RF soil packed at 1 g cm^{-3} was decreased by 31 and 38% respectively for GQW and SAR 50 treatments when the soils were compacted at 1.2 g cm^{-3} (Figure 5.8). When GQW and SAR 50 were applied to the BV soil packed at 1 g cm^{-3}, the cumulative Mg^{2+} removed after 10 PV were 599 and 940 mg kg^{-1} respectively. However, for the equivalent water treatments applied to ρ_b = 1.2 g cm^{-3} columns the cumulative Mg^{2+} was 34 and 39% lower, respectively.

A similar trend was observed for the cumulative K^+ removed in the leachate. Concentrations of cumulative K^+ removed after 10 PV of GQW and SAR 50 solutions were applied to the RF soil packed at 1 g cm^{-3} were 79 and 100 mg kg^{-1}. For the ρ_b = 1.2 g cm^{-3} columns, the total K^+ extracted was 48% lower for both water treatments. For the BV soil packed at 1 g cm^{-3}, the cumulative K^+ removed after 10 PV of GQW and SAR 50 was applied were 75 and 115 mg kg^{-1}, respectively. These values were 33 and 45% lower, respectively for the same treatments applied to the compacted (1.2 g cm^{-3}) columns.

5.3.4 Exchangeable cations and ESP

The exchange processes between the cations in the applied solution and cations on the soil exchange site were significantly affected by both soil pore size and soil depth. When the GQW solution was applied to the soils packed at 1 g cm^{-3}, exchangeable Ca^{2+} concentrations increased significantly ($P<0.05$) by 48, 45 and 35% in soil surface layer (0 – 1 cm) respectively for RF, GV and BV soils compared to the initial exchangeable Ca^{2+} concentrations (Table 5.2). A smaller increase (23, 25 and 19%) in exchangeable Ca^{2+} was observed in the bottom depth (7 – 8 cm). For soils compacted at 1.2 g cm^{-3} and treated with GQW, the exchangeable Ca^{2+} concentration increased at the soil surface by 25, 17 and 33% respectively for the RF, GV and BV soils compared to the initial concentrations.

Increases in exchangeable Ca^{2+} concentrations were associated with reductions in exchangeable concentrations of the other cations (Table 5-2). For example, exchangeable Na^+ concentrations in the soils packed at 1g cm^{-3} decreased significantly ($P<0.05$) by 85, 80 and 68% near the soil surface (0 – 1 cm) respectively for RF, GV and BV soil. However, in soils compacted at 1.2 g cm^{-3}, the reductions in exchangeable Na^+ at the surface were only 70, 71 and 30%, respectively. Exchangeable Mg^{2+} also decreased significantly ($P<0.05$) in the surface soil by 34 to 40% in the ρ_b = 1 g cm^{-3} treatment and by 15 – 25% for the ρ_b = 1.2 g cm^{-3} treatments. The exchangeable K^+ decreased at the by 25 – 43% for the ρ_b 1 g cm^{-3} treatment and by 11 – 21% for the ρ_b 1.2 g cm^{-3} treatments.

concentrations also were associated with reductions in exchangeable concentrations of the other cations (Ca^{2+}, Mg^{2+} and K^+).

For RF and BV soils packed at 1 g cm^{-3} and treated with SAR 10 and SAR 50, exchangeable Na^+ concentrations at the surface increased significantly ($P<0.05$) by 11 and 19 times for the RF soil and by 7.5 and 12 times for the BV soil compared to the initial exchangeable Na^+ concentration. At the bottom of the column (7 – 8 cm) exchangeable Na^+ concentrations increased by 5 and 11 times for RF soil and by 3.5 and 7.5 times for the BV soil. For the same soils treated with the same solutions (SAR 10 and SAR 50) but compacted at 1.2 g cm^{-3}, the exchangeable Na^+ concentrations increased at the surface by 8.5 to 14 times.

When solutions with high Na^+ concentrations were applied (SAR 10 and SAR 50) to the soils, exchangeable Na^+ concentrations increased significantly ($P<0.05$) throughout the soil columns (Table 5-2). For both ρ_b treatments, the increases were generally larger near the soil surface. The increases in exchangeable Na^+

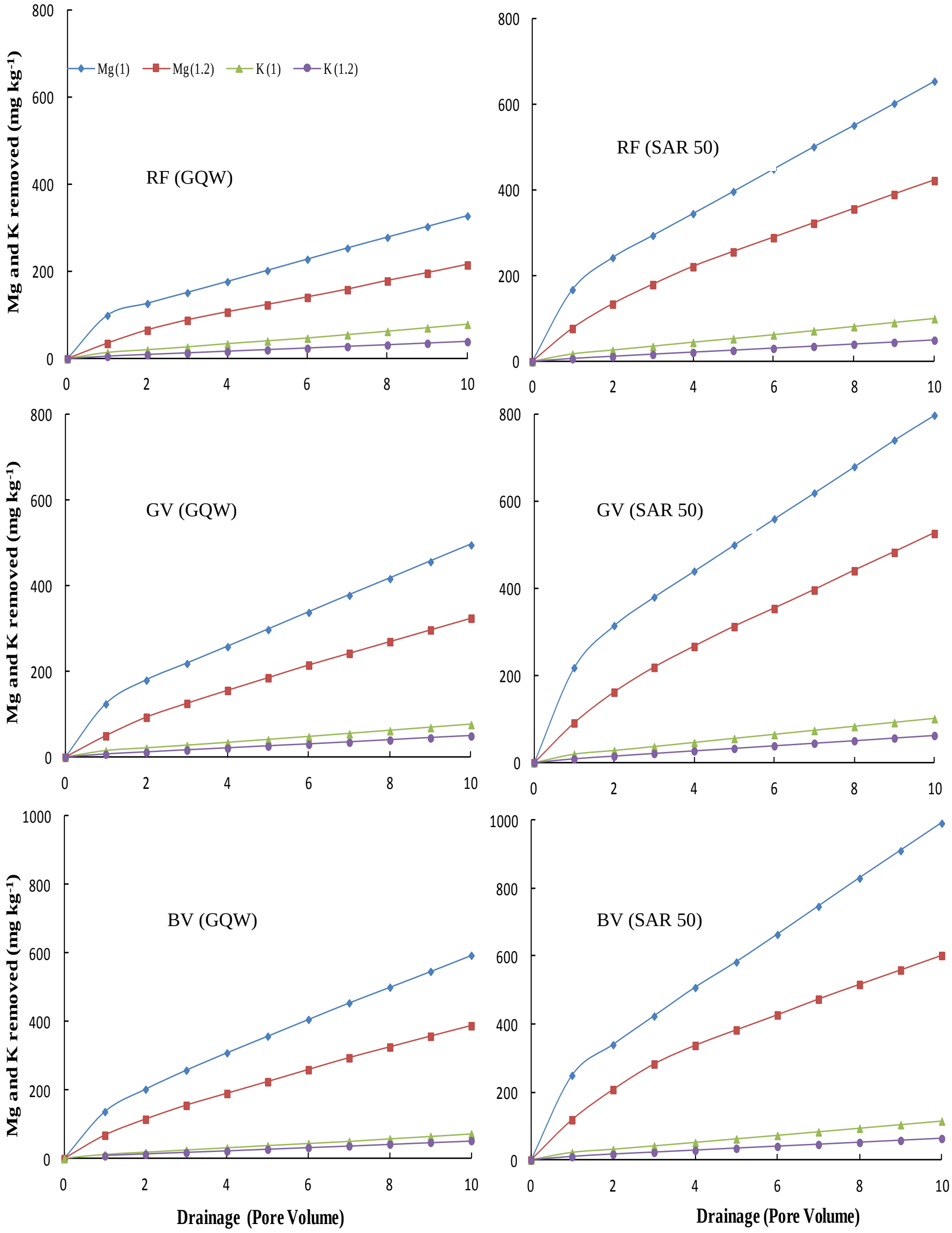


Figure 5-8 Cumulative extraction of Mg2+ and K+ with drainage from three soils (RF, GV and BV) packed at two bulk densities (1 and 1.2 g cm-3) and treated with GQW or SAR 50 solutions.

Table 5-2 Changes in exchangeable cations in the three soils packed at (1 and 1.2 g cm-3) saturated for 24 h and treated with 10 PVs of (GQW, SAR 10 and SAR 50) at three depths

Depth (cm)	ρ_b (g cm^{-3})	Na cmol (+) kg^{-1}				Ca cmol (+) kg^{-1}				Mg cmol (+) kg^{-1}				K cmol (+) kg^{-1}			
		Initial Ex.Na	GQW	SAR 10	SAR 50	Initial Ex.Ca	GQW	SAR 10	SAR 50	Initial Ex.Mg	GQW	SAR 10	SAR 50	Initial Ex.K	GQW	SAR 10	SAR 50
									RF soil								
0-1 cm	1	0.42^{a}	0.06^{bA}	5.0^{cA}	8.0^{dA}	9.5^{a}	14.1^{bA}	10.3^{cA}	6.2^{dA}	7.1^{a}	4.3^{bA}	2.2^{cA}	$3.3^{d A}$	0.80^{a}	0.60^{bA}	0.5^{cA}	0.5^{cA}
	1.2	0.42^{a}	0.12^{bB}	3.6^{cB}	6.0^{dB}	9.5^{a}	11.8^{bB}	10.0^{cA}	8.1^{dB}	7.1^{a}	6.3^{bB}	3.8^{cB}	4.3^{cB}	0.80^{a}	0.80^{aB}	0.6^{bB}	0.6^{bB}
4-5 cm	1	0.42^{a}	0.12^{bA}	3.0^{cA}	5.7^{dA}	9.5^{a}	11.2^{bcA}	11.3^{cA}	6.6^{dA}	7.1^{a}	6.9^{aA}	4.1^{bA}	5.2^{cA}	0.80^{a}	0.60^{bA}	0.4^{cA}	0.4^{cA}
	1.2	0.42^{a}	0.17^{bB}	1.5^{cB}	3.7^{dB}	9.5^{a}	10.3^{bB}	9.9^{cB}	8.1^{dB}	7.1^{a}	6.7^{abA}	6.3^{bB}	6.5^{bB}	0.80^{a}	0.70^{bB}	0.6^{cB}	0.6^{cB}
7-8 cm	1	0.42^{a}	0.11^{bA}	2.2^{cA}	5.0^{dA}	9.5^{a}	11.7^{bcA}	11.9^{cA}	8.3^{dA}	7.1^{a}	6.3^{bA}	4.2^{bA}	5.0^{cA}	0.80^{a}	0.70^{bA}	0.6^{cA}	0.6^{cA}
	1.2	0.42^{a}	0.17^{bB}	1.0^{cB}	3.0^{dB}	9.5^{a}	10.1^{abB}	10.4^{bB}	8.4^{cA}	7.1^{a}	6.7^{aA}	5.9^{bB}	6.1^{bB}	0.80^{a}	0.70^{bA}	0.6^{cA}	0.6^{cA}
									GV soil								
0-1 cm	1	1.30^{a}	0.24^{bA}	8.0^{cA}	14.0^{dA}	17.2^{a}	25.0^{bA}	21.1^{cA}	14.2^{dA}	15.4^{a}	10.1^{bA}	6.3^{cA}	7.4^{cA}	1.35^{a}	1.0^{bA}	0.7^{cA}	0.7^{cA}
	1.2	1.30^{a}	0.35^{bB}	6.0^{cB}	10.4^{dB}	17.2^{a}	22.2^{bB}	18.8^{dB}	15.7^{dA}	15.4^{a}	12.8^{bB}	9.7^{cB}	9.2^{cB}	1.35^{a}	1.1^{bA}	1.2^{bB}	1.1^{bB}
4-5 cm	1	1.30^{a}	0.73^{bA}	4.3^{cA}	10.0^{dA}	17.2^{a}	23.5^{bA}	20.4^{cA}	15.0^{dA}	15.4^{a}	11.0^{bA}	9.3^{cA}	9.4^{cA}	1.35^{a}	1.1^{bA}	1.0^{bA}	0.9^{bA}
	1.2	1.30^{a}	0.90^{bB}	2.5^{bB}	6.0^{dB}	17.2^{a}	20.3^{bB}	17.3^{aB}	16.1^{cB}	15.4^{a}	13.0^{bB}	13.4^{bB}	13.5^{bB}	1.35^{a}	1.2^{bA}	1.1^{bcA}	1.0^{cB}
7-8 cm	1	1.30^{a}	0.79^{bA}	3.1^{cA}	8.0^{dA}	17.2^{a}	22.3^{aA}	19.9^{bA}	15.1^{dA}	15.4^{a}	12.2^{bA}	10.9^{cA}	11.5^{cA}	1.35^{a}	1.2^{bA}	1.0^{cA}	1.1^{cA}
	1.2	1.30^{a}	1.12^{bB}	1.5^{cB}	4.0^{dB}	17.2^{a}	18.3^{bB}	17.5^{aB}	17.1^{aB}	15.4^{a}	15.5^{aB}	14.1^{bB}	14.1^{bB}	1.35^{a}	1.2^{bA}	1.1^{bcA}	1.0^{cB}
									BV soil								
0-1 cm	1	1.45^{a}	0.70^{bA}	11.2^{cA}	18.0^{dA}	21.9^{a}	29.9^{bA}	28.0^{bA}	19.0^{cA}	20.2^{a}	13.0^{bA}	5.0^{cA}	6.2^{cA}	2.94^{a}	2.4^{bA}	2.0^{cA}	2.1^{cA}
	1.2	1.45^{a}	1.10^{bB}	8.0^{cB}	13.0^{dB}	21.9^{a}	26.6^{bB}	26.0^{bB}	19.5^{cA}	20.2^{a}	18.1^{bB}	11.0^{cB}	12.5^{dB}	2.94^{a}	2.6^{bB}	2.4^{cB}	2.3^{cB}
4-5 cm	1	1.45^{a}	0.88^{bA}	6.0^{cA}	13.0^{dA}	21.9^{a}	27.1^{bA}	27.1^{bA}	19.0^{cA}	20.2^{a}	17.9^{bA}	10.0^{cA}	11.8^{cA}	2.94^{a}	2.5^{bA}	2.3^{cA}	2.2^{cA}
	1.2	1.45^{a}	1.1^{bB}	3.5^{cB}	8.0^{dB}	21.9^{a}	26.6^{bA}	23.9^{cB}	21.2^{dB}	20.2^{a}	20.1^{aB}	16.1^{bB}	16.7^{bB}	2.94^{a}	2.7^{bB}	2.5^{cB}	2.6^{cB}
7-8 cm	1	1.45^{a}	1.39^{bA}	5.1^{cA}	11.0^{dA}	21.9^{a}	26.3^{bA}	27.9^{bA}	20.6^{cA}	20.2^{a}	18.1^{bA}	11.2^{cA}	12.0^{cA}	2.94^{a}	2.6^{bA}	2.4^{cA}	2.4^{cA}
	1.2	1.45^{a}	1.50^{bB}	2.1^{cB}	6.0^{dB}	21.9^{a}	25.0^{bB}	21.9^{cB}	21.7^{cA}	20.2^{a}	19.9^{aA}	17.7^{bB}	18.0^{bB}	2.94^{a}	2.8^{bB}	2.6^{cb}	2.6^{cB}

Small letters (*a, b, c and d*) compare between treatments and capital letters *(A, B and C*) compare between depths

The ESP of soils packed at 1 g cm^{-3} and treated with SAR 10 and SAR 50 was significantly ($P<0.05$) high than when the columns were compacted at 1.2 g cm^{-3} (Table 5.3). For example, the surface ESP of RF and BV soils treated with SAR 50 and packed at 1 g cm^{-3} was approximatly1.3 times higher than when compacted at 1.2 g cm^{-3}.

Table 5-3 ESP of the RF, GV and BV soils treated with GQW, SAR 10 or SAR 50 at three depths (0-1, 4-5 and 7-8 cm)

Depth (cm)	ρ_b (g cm^{-3})	GQW	SAR 10	SAR 50
		RF soil		
0-1 cm	1	0.31^{aA}	26.06^{bA}	42.12^{cA}
	1.2	0.63^{aB}	19.18^{bB}	31.50^{cB}
4-5 cm	1	0.62^{aA}	15.79^{bA}	30.01^{cA}
	1.2	0.89^{aB}	7.89^{bB}	19.21^{cB}
7-8 cm	1	0.57^{aA}	11.58^{bA}	26.32^{cA}
	1.2	0.89^{aB}	5.26^{bB}	15.47^{cB}
		GV soil		
0-1 cm	1	0.65^{aA}	21.68^{bA}	38.90^{cA}
	1.2	0.96^{aB}	16.46^{bB}	27.54^{cB}
4-5 cm	1	2.00^{aA}	11.54^{bA}	27.43^{cA}
	1.2	2.46^{aA}	6.60^{bB}	16.09^{cB}
7-8 cm	1	2.16^{aA}	8.72^{bA}	21.44^{cA}
	1.2	3.07^{aB}	4.16^{bB}	10.97^{cB}
		BV soil		
0-1 cm	1	1.45^{aA}	23.00^{bA}	37.83^{cA}
	1.2	2.29^{aB}	16.66^{bB}	27.30^{cB}
4-5 cm	1	1.83^{aA}	12.66^{bA}	27.03^{cA}
	1.2	2.29^{aB}	7.41^{bB}	16.66^{cB}
7-8 cm	1	2.89^{aA}	10.58^{bA}	22.08^{cA}
	1.2	3.12^{aB}	4.33^{bB}	12.50^{cB}

Small letters (*a, b and c*) compare between treatments and capital letters (*A, B and C*) compare between depths

5.4 Discussion

5.4.1 Effect of ion exchange rate on soil Ks

Soils packed at 1 g cm^{-3} leached significantly ($P<0.05$) more cations within 1-2 PV compared to those compacted at 1.2 g cm^{-3}. The discharge of high concentrations of cations in low ρ_b soil during the first PV of leaching confirmed leaching of soluble cations from soil macropores and mesopores which responded to fast water flow within preferential flow paths (Figure 5.9). After 1 PV, the concentration of cations leached decreased with a significant decrease in Na^+ concentration (dominant cation in SAR solutions) demonstrating exchange between Na^+ in the applied solution and soil cations on exchange sites (Jalali and Merrikhpour, 2007). As water flow within soil under saturated condition is mainly controlled by macropores (Hillel 2004). This could signify the start of exchange processes within preferential flow paths (macropores). It follows that significant reductions in soil K_s for all soils studied occurred at the start of the leaching processes. However, as leaching processes continued after 3 - 6 PV had been applied but no significant changes in soil K_s were observed during the later stages of the leaching (Figure 5-9). The exchanges processes during later leaching mainly occur within smaller pores (micropores) and contributed only a small proportion of total soil K_s (Hillel 2004).

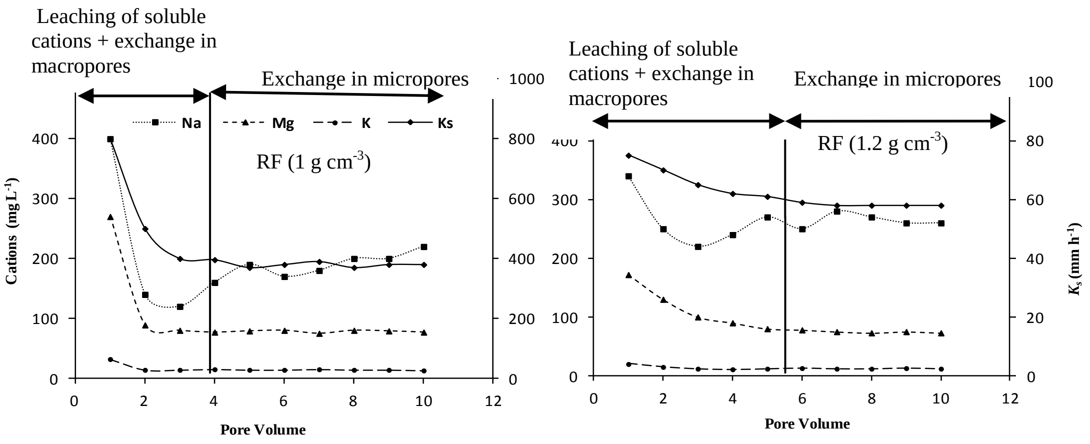


Figure 5-9 Soil Ks (Right axis)along with Na, Mg and K concentration (Left axis) of the leachate from soil columns packed at two densities showing stages of ion exchange in different pore sizes.

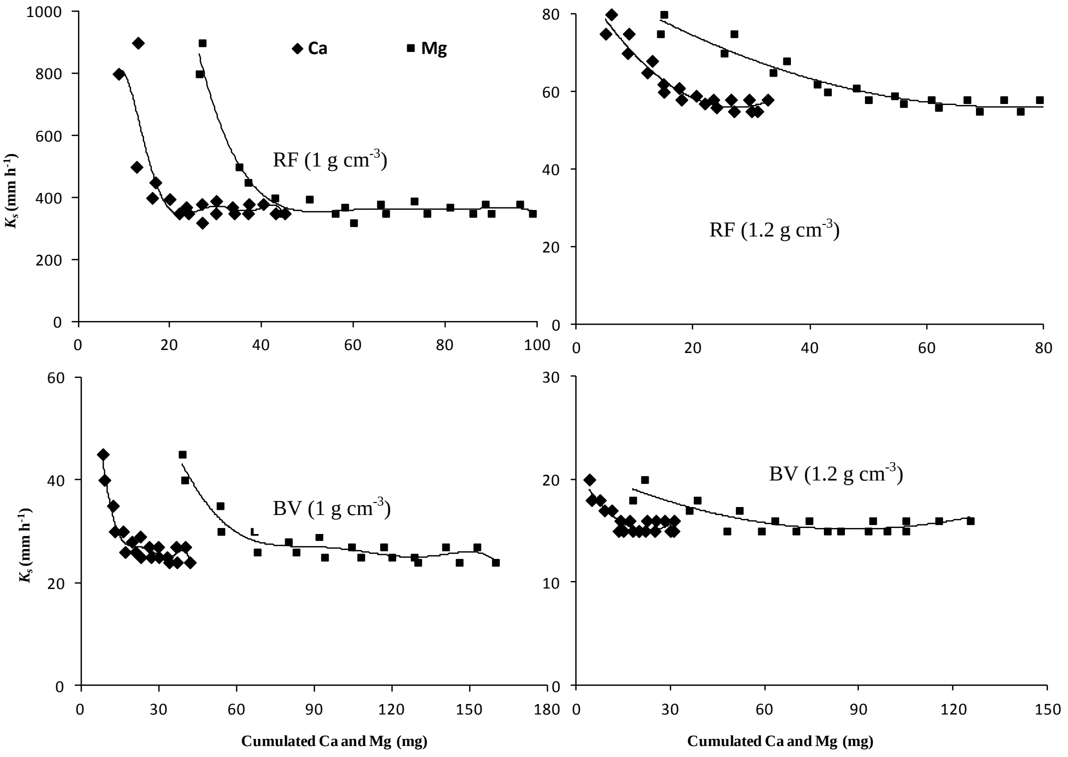


Figure 5-10 Relation between soil Ks and cumulative ion concentrations in the leachate of RF and BV soils packed at ρb (1 or 1.2 g cm-3) and treated with SAR 50 solution.

Changes in soil K_s and the cumulated ion exchange (Ca^{2+} and Mg^{2+}) for RF and BV soils showed that soil K_s reductions were larger for low ρ_b soils, particularly at the start of leaching (Figure 5.10). The correlation analyses indicated that soil K_s was negatively correlated with the cumulated cations exchanged in all treatments. However, the reductions in soil K_s the ρ_b = 1 g cm^{-3} treatment were more highly correlated (R^2 = - 0.90 and - 0.74 respectively for RF and BV soils) than the ρ_b = 1.2 g cm^{-3} treatments (R^2 = - 0.68 and – 0.63, respectively). This suggests that K_s for the RF soil is slightly more affected by the rate of ion exchange than the BV soil, possibly due to the swelling properties of the BV soil where rich concentration of 2:1 clays swell as they come into contact with water clays (McNeal and Colemam, 1966;

Yaron and Thomas, 1968). Since the RF soil is a non-swelling soil, it could be suggested that the main mechanism for soil K_s reductions with increasing the rate of ion exchange was pore blockage by clay dispersion.

In compacted soil, the higher proportion of micropores within soil aggregates caused slower leaching of soluble cations from the soil and more drainage was needed to leach the soluble cations. This suggests that cations are located within aggregates or micropores may take longer to exchange into the percolating solution (Figure 5.9). The delayed soluble cation leaching from micropores slowed the exchange processes in the compacted soil. Significant reductions in K_s were observed during early stage leaching with saline-sodic solutions (up to 5 PV), presumably due to the impact of exchangeable Na^+ on the structure of soil macro-aggregates and macropores. Unlike low ρ_b soil, soil K_s of the compacted soil kept decreasing slightly during later stage leaching due to exchange processes in soil micropores. This confirms that Na^+ exchange within micropores has some effect on K_s.

The reduction in K_s values were more pronounced in BV and GV soil compared to RF soil due to clay mineralogy. The RF soil contains iron oxides that strengthen soil aggregates (Deshpande et al., 1968; Goldberg et al., 1988), while the GV and BV soils contain a high proportion of 2:1 clays that swell and reduce the pore size particularly under sodic conditions (McNeal and Colemam, 1966; Yaron and Thomas, 1968).

5.4.2 Effect of soil pore size distribution on soil chemical equilibrium

Cation exchange occurred further and to a larger extent in the low ρ_b soil compared to compacted soil. This is most likely due to preferential flow through macropores under saturated conditions and a larger porosity allowing a greater volume of solution to pass through soil columns (Reading et al. 2012).

The increase in exchangeable Na^+ and reductions in other exchangeable soil cations with leaching was most evident near the soil surface and decreased with depth. This result is attributed to replacement of cations such as Ca^{2+}, Mg^{2+} and K^+ held at exchange sites by Na^+. Increased Na^+ on the clay exchange sites reduced soil K_s, thus increasing the contact time between the soil and the water applied. This would allow for more complete exchange of cations at sites which increased ESP near the soil surface. Subsequently, as water infiltration progressed, Ca^{2+}, Mg^{2+} and K^+ were transported in the soil water thereby increasing their relative concentration at greater depths compared with Na^+. As a result, Ca^{2+}, Mg^{2+} and K^+ competed with Na^+ for exchange sites at those depths and yielded a relatively lower ESP values compared with that recorded near the surface. These results agree closely with those reported in earlier studies (Jalali and Merrikhpour, 2006 and Abu-Sharar et al. 1987).

Different trends were observed for soil cation exchange processes after leaching with 10 PV of water. The increasing Ca^{2+} concentration in the soil solution when GQW was applied led to a considerable reduction in exchangeable Na^+ concentrations particularly at soil surface for all soils. This confirmed the high exchange rate between Ca^{2+} in soil solution and Na^+ on exchange sites likely due to Na^+ cations are less held on exchange sites compared to other cations due to large hydrated radius and low valency of Na^+. However, RF soil exhibited larger reductions in exchangeable Na^+ compared to GV and BV soils. That may be attributed to higher proportion of soil macropores in RF soil compared to GV and BV soils that improved the exchange processes.

The exchange rate of K^+ was very slow in BV and GV soil compared to RF soil when saline-sodic solutions were applied. This may be attributed to K^+ cations having a smaller hydrated radius allowing them to more readily enter the interlayer space between the clay sheets, particular in GV and BV soils that are rich in 1:2 clay minerals (Sparks, 2003). In addition, a high exchange rate of Mg^{2+} in the three soils was noted compared to the other cations. This could be related to a high concentration of exchangeable Mg^{2+} on the exchange sites and the large hydrated radius of Mg^{+2} that may accelerate the exchange rate despite double positive charge (Sparks, 2003). The behaviour of exchangeable soil cation, clay mineralogy and soil pore size may provide an explanation for RF approaching chemical equilibrium faster than GV and BV soils.

5.4.3 EC as an indicator for chemical equilibrium and Ks steady state

The EC of the leachate was initially high during the first few pore volumes of drainage and then decreased to a value similar to the EC of the percolating solutions. Low ρ_b soils reached a stable EC with less drainage than the compacted soil, due to the higher rate of soil soluble salt flushing as already discussed. Reading et al. (2012) and Shackelford et al. (1999) have suggested that when the EC of the leachate is stable this is a good indicator of chemical equilibrium within column. However, the exchangeable cation measurements suggested that complete ionic exchange equilibrium was not approached for any of the soils packed at either ρ_b studied. The results presented here suggest that the steady state EC during leaching is more appropriately a good sign of complete soil soluble salt leaching and cation exchanges within only the dominating soil pore flow paths. Other studies have also showed that the EC may reach a steady state even though chemical equilibrium for the whole soil was not reached (Misra and Sivongxay, 2008; and Jalali and Ranjbar, 2009; Wieck 2012). Thus, EC cannot be considered as an indicator for the chemical equilibrium state of the whole soil.

K_s values and EC in the leachate followed a similar trend where they reached steady states within the 10 PV of leaching. The K_s values were initially high then declined as the EC of the leachate began to decrease towards the steady state, particularly in the high SAR 50 treatment (Figure 5.11). When soil K_s approached the steady state, the EC of the leachate also approached steady state with a slight increase in the leachate EC as compared to the EC of applied solution due to cations such as Mg^{2+} and K^+ continuing to discharge in the leachate. This indicates the continuance of exchange processes involving Mg^{2+} and K^+ which were not added in the applied solutions. These results suggest that the, exchange processes carried out within soil micropores within soil aggregates have relatively a small effect on soil bulk water movement.

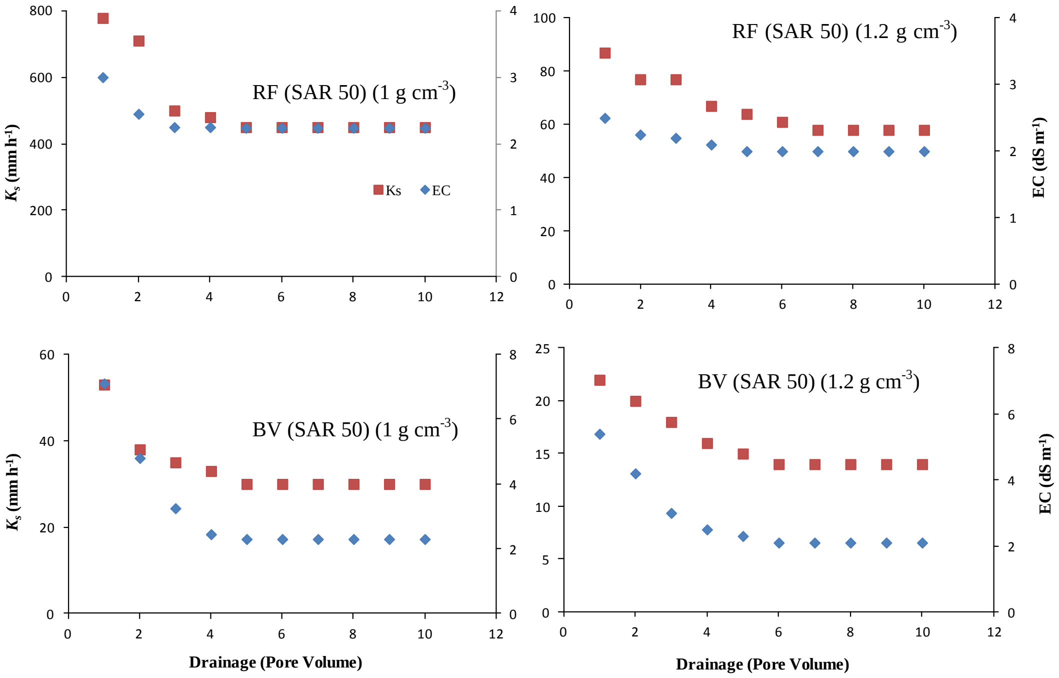


Figure 5-11 Changes in Ks (left axis) and with EC (Right axis) during leaching of two soils (RF and BV) at different bulk density (1 and 1.2 g cm-3) and treated with SAR 50 solutions.

5.5 Conclusion

Soils packed at low ρ_b (1 g cm^{-3}) were capable of discharging greater concentrations of soluble salts during 1 PV of leaching with saline-sodic solutions. The fast flushing of soluble salts was due to the higher percentage of soil macropores that allowed faster water and soluble ion movement within the column. After 1 PV of drainage the concentrations of Na^+ in the leachate decreased significantly indicating the replacement of Na^+ with exchangeable soil cation between soil aggregates (macropores). The effect of exchangeable Na^+ on preferential flow paths between soil aggregates was the reason for significant reductions in soil K_s during the early stage of leaching. However, the rate of ion exchange was better correlated with changes in soil K_s in non-swelling soil (RF) soil compared to swelling soil (BV). That may indicate some reduction in soil K_s in the BV soil was associated with the swelling properties of the dominant clays (2:1 clays) in the BV soil. During later stage leaching (> 3 PV) no significant changes in Na^+ and K_s were observed signifying that exchange processes between soil cations and Na^+ of applied solution were likely occurring within micropores and this exchange had less effect on water movement under saturation conditions.

In compacted soil (ρ_b=1.2 g cm^{-3}) with a higher proportion of micropores, less flushing of soluble salts was observed after 1 PV of drainage. However; it required 5 PV to produce K_s steady state suggesting that leaching of soluble salts within micropores is a slower process in compacted soils. Similar to the low ρ_b soil, significant reductions in soil K_s were obtained within the first 5 PV of leaching due to ionic exchange between soil cations and Na^+ in the applied solution. However, the reductions in soil K_s were significantly less than the reductions observed in the lower ρ_b soil due to a smaller proportion of soil macropores. In contrast to low ρ_b soil, soil

K_s continued to decrease for up to 8 PV of drainage most likely due to the high proportion of soil micropores in compacted soil.

The ESP was significantly higher near the soil surface compared to deeper depths. This was due to a longer contact time with the applied solution. Also soil cations (Ca^{2+}, Mg^{2+} and K^+) exchanged with Na^+ moved with infiltrated water to lower depths where their concentration increased relative to Na^+.

Chapter 6: The Fate of Dispersed Clay and Interaction with Soil Pore Size

6.1 Introduction

The saturated hydraulic conductivity K_s of soils is an important consideration in soil management as it influences critical processes that occur in the soil-plant system. The K_s of soils depends on soil porosity and pore size distribution. In irrigated soils, these two parameters are influenced by the quality of irrigation water used (Speirs et al. 2011; Alaoui et al. 2001). Irrigation with saline-sodic water can result in accumulation of Na^+ in the soil profile which can become sodic (Minhas et al. 1998)). Sodic soils are characterised by poor structure and low aggregate stability because of swelling and closures of inter-aggregate pores, and the dispersion of fine clay particles (Shainberg and Letey 1984). Soil dispersion blocks soil pores and reduces soil permeability to air and water (Rengasamy and Olsson, 1991). This leads to low infiltration and poor aeration of the rooting zone and impairs root growth and plant establishment (Van Hoorn and Van Alphen, 1994).

Irrigation with saline-sodic water decreases soil K_s because it increases soil ESP (Curtin et al. 1994). It is generally accepted that values of ESP above 15% have adverse effects on soil structure; aggregate stability and soil pore geometry (Mihas and Sharma 1986). The effects of Na^+ on soil structural stability occur through the following processes (after Sumner, 1993; Aringhieri and Giachetti, 2001; Mace and Amrhein, 2001): (i) dispersion and subsequent migration of dispersed clay sediments that block soil pores, or (ii) clay swelling which narrows soil pores and results in clay dispersion. The potential for structural degradation due to Na-rich irrigation water is more severe at high values of ESP and when the water applied has an EC less than a threshold EC (TEC) which leads to clay swelling and dispersion causing pore blockage (Sumner, 1993).

Minhas and Sharma (1986) examined the effects of applying saline-sodic water with different SAR (range: 5 to 45) and electrolyte concentrations (range: 15 to 90 meq L^{-1}) followed by the application of distilled water to two soils (a sandy loam and a clay loam). The authors reported a significant reduction in hydraulic conductivity which was greater in the clay loam soil due to the relatively higher clay content, and with increasing SAR and decreasing TEC. The study attributed the reduction in K_s to dispersion of clay particles and subsequent pore clogging. A later study (Mace and Amrhein, 2001) examined lower SAR values (range: 1 to 8) and a wider range of electrolyte concentrations (from 0 to 100 $mmol_c$ L^{-1}) on hydraulic conductivity, clay dispersion and pore blockage. The initial water application was followed by gypsum and sulphuric acid applications to the soil surface and subsequent leaching with distilled water. The concentration of clay in the drainage water was used to quantify clay dispersion. Hydraulic conductivity was reduced at all SARs after applying distilled the water. Clay dispersion, pore blockage and consequently K_s under sodic conditions is also significantly affected by clay mineralogy. In this regard, reductions in soil K_s occur to greater extent in montmorillonitic soils compared with kaolinitic soils (Das and Datta, 1986). These studies all consider K_s as an indicator of clay dispersion and pore blockage. However, few studies have investigated the effect of pore size on the movement of the dispersed sediment within the soil profile and the potential of zones of pore blockage. This has important implication for the management of irrigated soils as the clay migration and zones of blockage will impact on the potential for surface sealing, crusting and hardsetting behaviour. Hence, the objective of this chapter is to evaluate the effect of pore size on the migration of dispersed clay and zones of pore blockage in soils with contrasting mineralogy.

6.2 Material and methods

6.2.1 Preparation of soil cores and leachate analyses

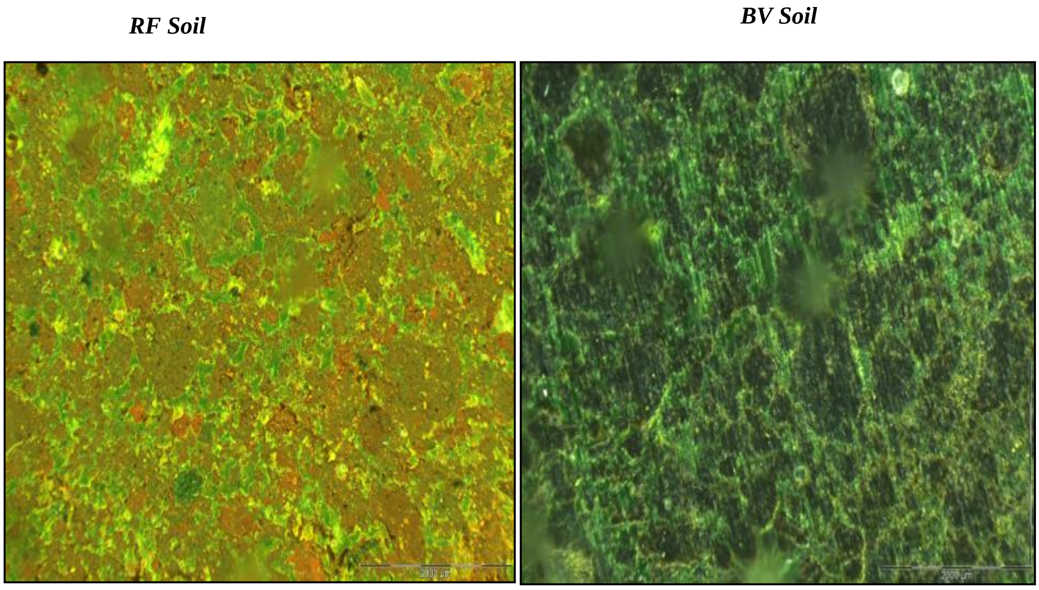
The same three soils (RF, GV and BV) used in the earlier chapters were prepared and packed into 8 cm columns at $\rho_{b} =$ 1 or 1.2 g cm^{-3} following the method outlined in chapter 3. Table 3.1 shows the properties of the soils used in this study. Three solutions were prepared using the procedure outlined in section 3.3.2 to simulate (a) good quality water (SAR = 0.11, EC = 1 dS m^{-1}), (b) saline-sodic water (SAR = 10, EC = 2 dS m^{-1}) and (c) highly saline-sodic water (SAR = 50, EC = 2 dS m^{-1}). Soil columns were wetted from the bottom and equilibrated for 24 h in the solutions. Five replicates for each treatment were conducted. After equilibration, a total of 10 pore volumes (PV) of each solution were infiltrated though the soil columns. The soil columns were then covered and drained before applying a constant 2 cm head of rain water (RW) to the soil surface. The RW was applied until four PV of leachate had drained from the columns. The leachate was separated at each PV and analysed for sediment concentration using the method described in section 4.3.2 and also for EC and cation concentration (Na^{+}, Ca^{2+}, Mg^{2+} and K^{+}) using the method described in section 3.3.6. After drainage, two replicates of the soil columns were sectioned into three depths (0 – 1, 3 – 4 and 5 – 6 cm) and the exchangeable cations (Na^{+}, Ca^{2+}, Mg^{2+} and K^{+}) were measured using the method repeated in Section 3.3.6.

6.2.2 Soil cores impregnation and soil block preparation

A total of 52 columns were impregnated with resin to evaluate pore size distribution. One column of each soil packed at 1.0 and 1.2 g cm^{-3} and where no water had been applied were prepared. Similarly, two replicates of each water quality treatment. (GQW, SAR 10 and SAR 50) applied on the RF and BV soils (as used in Chapter 5) were prepared. Two replicates of the same soil, density and water treatments but after applying the RW were also prepared. The soil columns were prepared and impregnated using the method outlined in section 5.2.1. The soil columns were subsequently sectioned and prepared into 20 mm soil depth layers (0, 2, 4, 6 and 8 cm) using the method described in section 5.2.1. The surfaces of the soil sections were imaged (Figure 6-1) under a microscope set at a magnification of 65 X and analysed using TBitmap software (NCEA staff, 2013) (Figure 6-2). The image analysis involved:

(i) Resin colour analysis to identify the pixels in the image as pores;
(ii) Flood fill algorithm to determine the connected components in the image (i.e. the number and size of each pores).

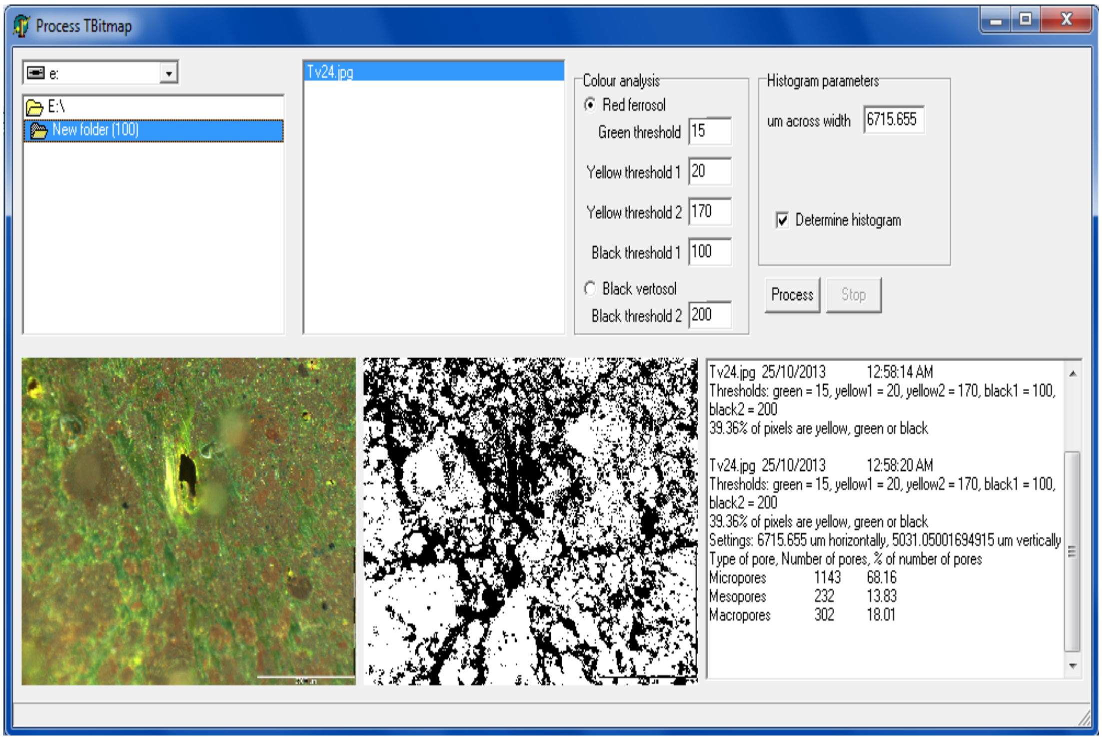
The pore size distribution was classified using the Kay (1997) method in which pores are classed as either (a) micropores (< 0.32 µm); (b) mesopores (0.2-30 µm) and (c) macropores (> 30 µm).



RF Soil

BV Soil

Figure 6-1 Example image of soil surface showing soil pores (left image for RF and right image for BV) where green area represents soil pores and grey and black areas represent soil matrix.



● Red ferrosol

Green threshold 15

Yellow threshold 1 20

Yellow threshold 2 170

Black threshold 1 100

○ Black vertosol

Black threshold 2 200

um across width 6715.655

☑ Determine histogram

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Tv24.jpg  25/10/2013        12:58:14 AM
Thresholds: green = 15, yellow1 = 20, yellow2 = 170, black1 = 100,
black2 = 200
39.36% of pixels are yellow, green or black

Tv24.jpg  25/10/2013        12:58:20 AM
Thresholds: green = 15, yellow1 = 20, yellow2 = 170, black1 = 100,
black2 = 200
39.36% of pixels are yellow, green or black
Settings: 6715.655 um horizontally, 5031.05001694915 um vertically
Type of pore, Number of pores, % of number of pores
Micropores        1143      68.16
Mesopores         232       13.83
Macropores        302       18.01
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Figure 6-2 Example analysis screen using TBitmap software for a surface image of RF soil.

6.2.3 Determination of the number of images required to characterise porosity

High resolution images of soil pores are preferred due to their ability to identify smaller soil pore sizes (Borges et al. 2012). However, the higher the resolution the smaller the area that is imaged and the larger number of images that will be required to adequately characterise the porosity. Hence, the minimum number of representative areas sample images required will differ from one soil to another depending on the variation in soil physical properties (Baveye et al. 2002; VandenBygaart and Protz, 1999). This experiment evaluated image variance for two different depths (0 and 40 mm) on the RF and BV soils packed at 1.0 g cm^{-3} and treated with either GQW or SAR 50. The standard deviation for each number of samples was measured by randomly selecting a subset of the total number of images. Figure 6.3 shows that the standard deviation did not change significantly when at least 14 images were used for the RF soil and 10 were used for the BV soil. This number of images represented 24% and 20.6 % of the total surface area of the soil block, respectively. Hence, in subsequent analyses of porosity, 16 images were used for each soil representing 27.5% of the total surface area of the soil block.

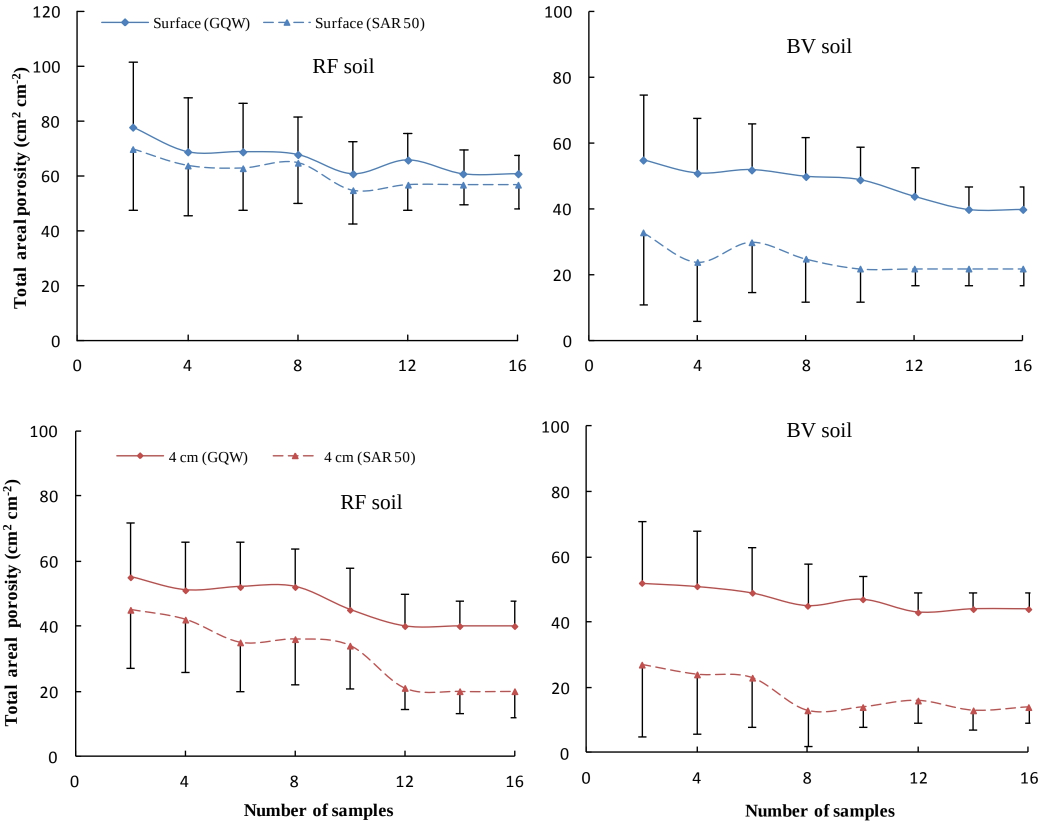


Figure 6-3 Standard deviation between selected samples of soil images for RF and BV soils treated with GQW and SAR 50 treatments at depths of 0 and 4 cm.

6.3 Results

6.3.1 Saturated hydraulic conductivity (Ks)

The soil K_s declined significantly ($P<0.05$) following application of RW particularly for columns that had been previously treated with SAR 10 or SAR 50 saline-sodic solutions (Figure 6-4). Maximum K_s reductions occurred after leaching with two PV of RW for low ρ_b soil compared to three PV in the more compacted soils. Smaller reductions were noted for soils treated with GQW (Figure 6.4). Reductions in soil K_s

were larger in soils packed at 1 g cm^{-3} compared to those compacted at 1.2 g cm^{-3} (Figure 6.4).

For the RF soil packed at 1 g cm^{-3} and treated previously with SAR 50 solution, the final K_s after RW application was only 14% of the K_s of the column treated with GQW. For the same soil treatment compacted at 1.2 g cm^{-3}, the final K_s was 3% of the GQW column. When RW was applied to the RF soil previously treated with SAR 10 water K_s decreased by about 40% and 30% compared with GQW treated-soils packed at 1.0 and 1.2 g cm^{-3}, respectively.

For the GV soil, the final K_s values decreased significantly by 97% for the 1 g cm^{-3} column and by 70% for the 1.2 g cm^{-3} column previously treated with SAR 50. These reductions were 62 and 50 % for the GV soil treated with SAR 10. Larger reductions in K_s were noted with application of RW to the BV soil treated with SAR 50 water. The final K_s decreased by 97% and 88%, respectively for the low and high density soils compared with columns treated with GQW.

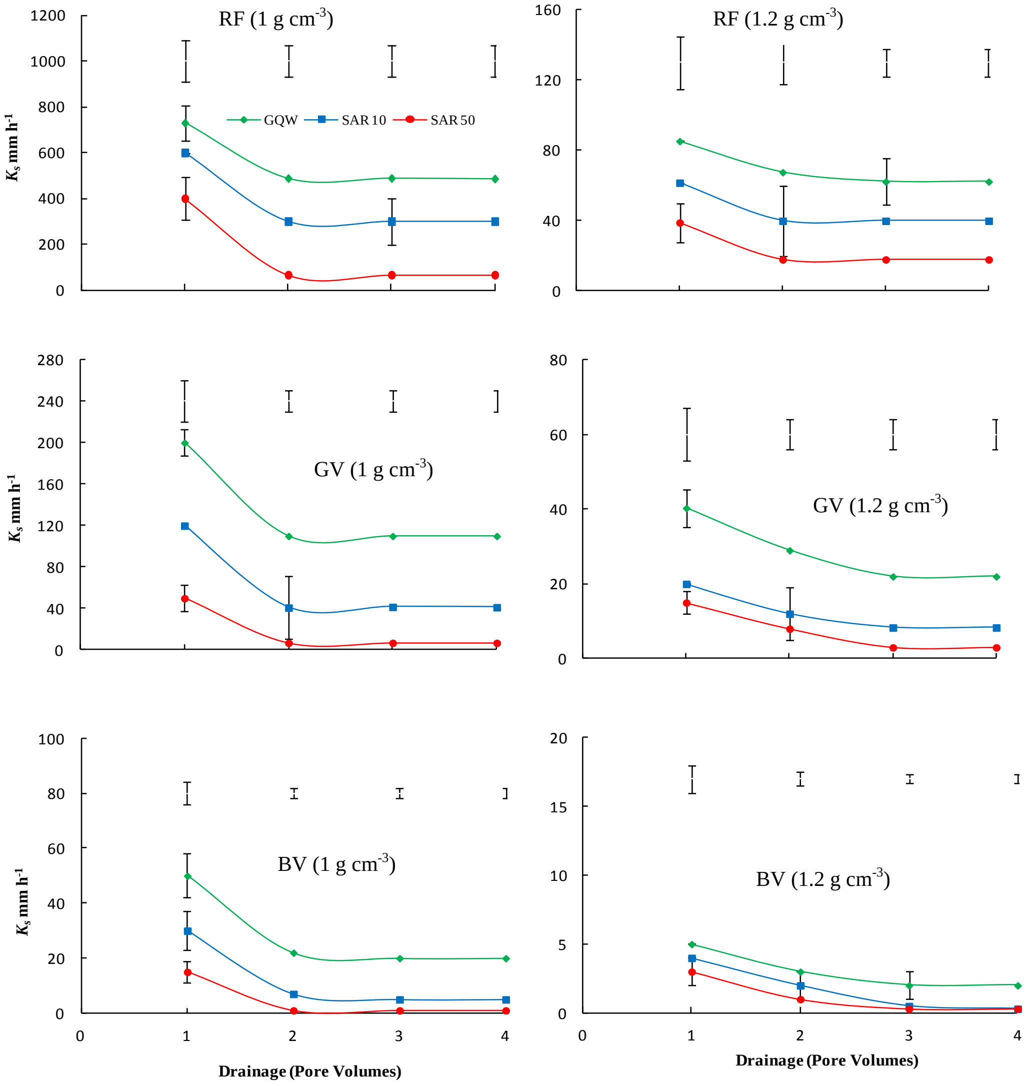


Figure 6-4 Ks of three clay soils (RF, GV and BV soils) packed at two bulk densities (1 and 1.2 g cm-3) and treated with RW after irrigation with different water qualities (GQW, SAR 10 and SAR 50). Bars on the graph are LSD (α = 0.05) between treatments and bars plotted on curve are LSD within treatments.

6.3.2 Leachate EC

After one PV of drainage the EC of the leachate from all soil columns irrigated with EC = 2 dS m^{-1} solutions was significantly higher than the leachate EC from columns irrigated with GQW (EC = 1 dS m^{-1}) (Figure 6-5). The EC of the leachate reached a steady state for soil packed at 1 g cm^{-1} after 2 PVs of drainage. However, for compacted soil the EC continued to decrease until 3 PV had drained (Figure 6.5).
The leachate EC from the lower density soils was significantly higher at the start of leaching compared to the higher density soils. For example, as RW was applied to the RF soil packed at 1 g cm^{-3} and previously irrigated with GQW, the leachate EC was 0.46 dS m^{-1} after 1 PV but then decreased by 80% and 90% after 2 and 4 PV, respectively. For the same soil packed at 1.2 g cm^{-3} and previously irrigated with GQW, the EC value was 0.38 dS m^{-1} after 1 PV. This decreased by 60 % after 2 PV and by 77 % after 4 PV. Similarly when RW was applied to the BV soil packed at 1 g cm^{-3} and previously treated with GQW, the EC was 1 dS m^{-1} after 1 PV but decreased by 80 and 90 % after 2 and 4 PV, respectively. The leachate EC was 0.8 after 1 PV for the same treatment applied to BV soil compacted at 1.2 g cm^{-1} but decreased by 68 and 86% after 2 and 4 PV, respectively.

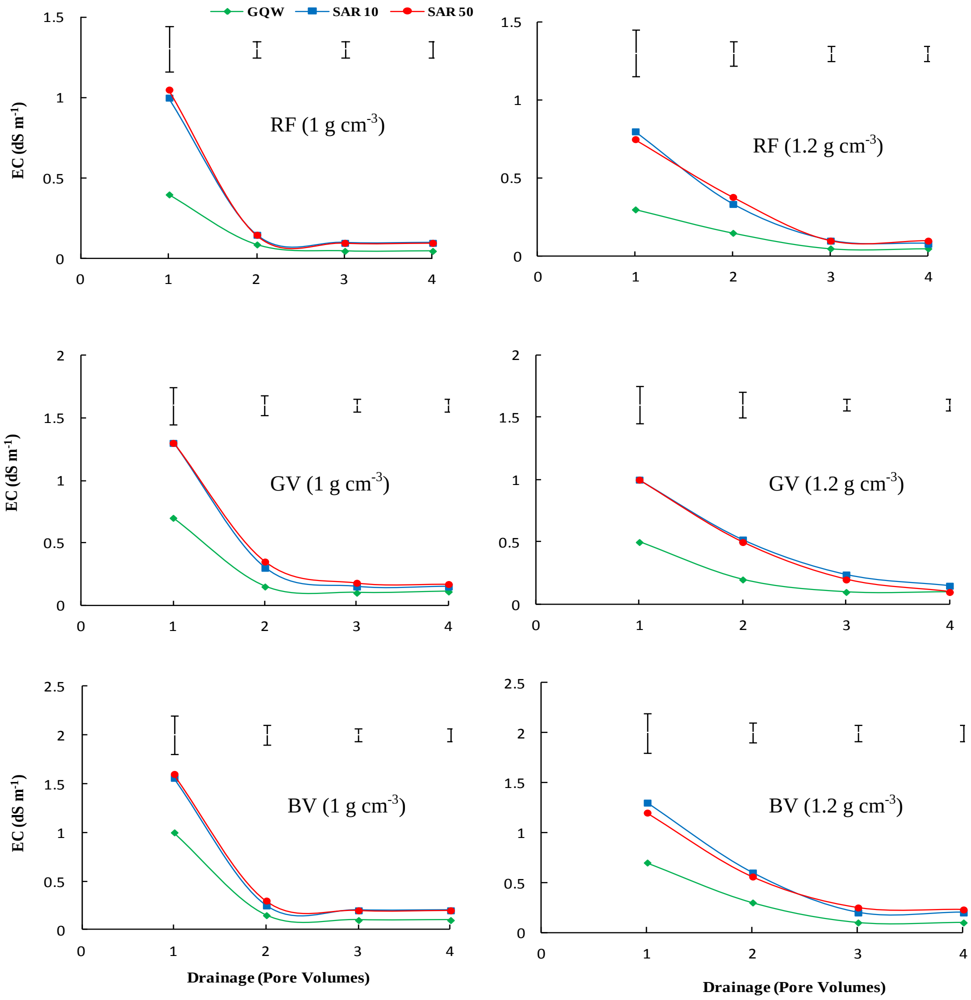


Figure 6-5 Leachate EC (dS m-1) for three clay soils (RF, GV and BV) packed at two densities (1 and 1.2 g cm-3) and treated with different water qualities (GQW, SAR 10 and SAR 50) after applying RW. Bars placed on the graph are LSD (α = 0.05) between treatments.

6.3.3 Dispersed clay in the leachate

The dispersed clay concentration of the leachate generally increased with increasing SAR of the previously applied water (Figure 6-6). For instance, the dispersed clay concentration after applying RW to the RF soil packed at 1 g cm^{-3} and treated earlier with GQW was 0.36 g L^{-1} after 1 PV but was 13 times higher for the same soil treated with SAR 50 water. In addition, the cumulative dispersed clay concentration after 4 PV was 20 times greater for the SAR 50 treatment compared to the GQW treatment.

The dispersed clay concentration in the leachate from the BV soil was generally higher than leachate from the other soils. For RW applied to the BV soil packed at 1 g cm^{-3} and treated previously with GQW water, the dispersed clay concentration in the leachate was 1 g L^{-1} after 1 PV but 5 times higher when treated with SAR 50 water. However, there was no significant difference between the two treatments after 4 PV (Figure 6-6).

The dispersed clay concentration in the leachate was significantly lower for soils packed at 1.2 g cm^{-3} compared to soils packed at 1 g cm^{-3}. For example, cumulative dispersed clay concentration in the leachate from the RF columns packed at 1.2 g cm^{-3} and treated with SAR 10 and SAR 50 water was 53 and 86% lower, respectively, than leachate from columns packed at 1 g cm^{-3}. Similarly, the clay concentration in the leachate from BV soil packed at 1.2 g cm^{-3} after treating with SAR 10 and SAR 50 water was 45 and 59 % lower than the leachate from columns packed at 1 g cm^{-3} (Figure 6-6).

6.3.4 Cations in the leachate

The leachate cation concentration was generally lower for the GQW treatment compared to the higher EC and SAR treated soil columns. Cation (Na^+, Ca^{2+}, Mg^{2+} and K^+) concentrations in the leachate generally decreased with increasing drainage (Figures 6-7 to 7-10). There was generally no significant difference in the leachate concentration after 2 PVs for the low density columns and after 3 PVs for the high density columns. The concentrations of leached cations were significantly higher for lower density soil compared to compacted soils after 1 PV while the compacted soils generally had higher leachate concentrations as leaching continued until 4 PV.

The Na^+ and Ca^{2+} concentration in the leachate after 1 PV was generally significantly higher with increasing SAR of the solution applied to the soil columns (Figure 6-7 and 6-8). However, there was no significant difference in Mg^{2+} and K^+ between the SAR treatments for each soil and packing density (Figure 6-9 and 6-10). The K^+ concentrations in the leachate were low compared to the other cations.

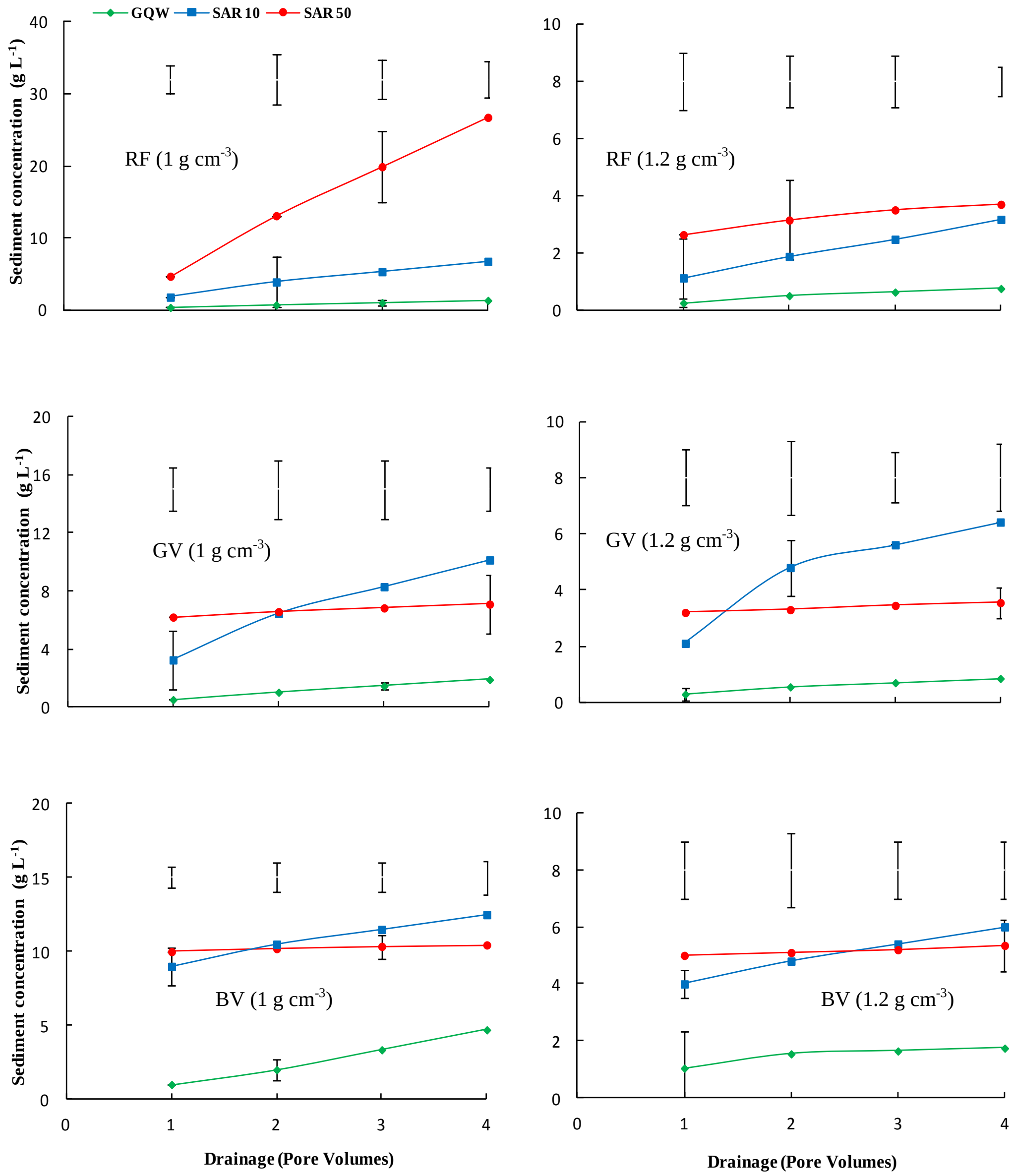


Figure 6-6 Cumulative dispersed clay concentrations in the leachate (g L-1) for three clay soils (RF, GV and BV soils) packed at two densities (1 and 1.2 g cm-3) and treated with different water qualities (GQW, SAR 10, SAR 50) with pore volume after applying RW. Bars placed on the graph are LSD (α = 0.05) between treatments and bars plotted on curve are LSD within treatments.

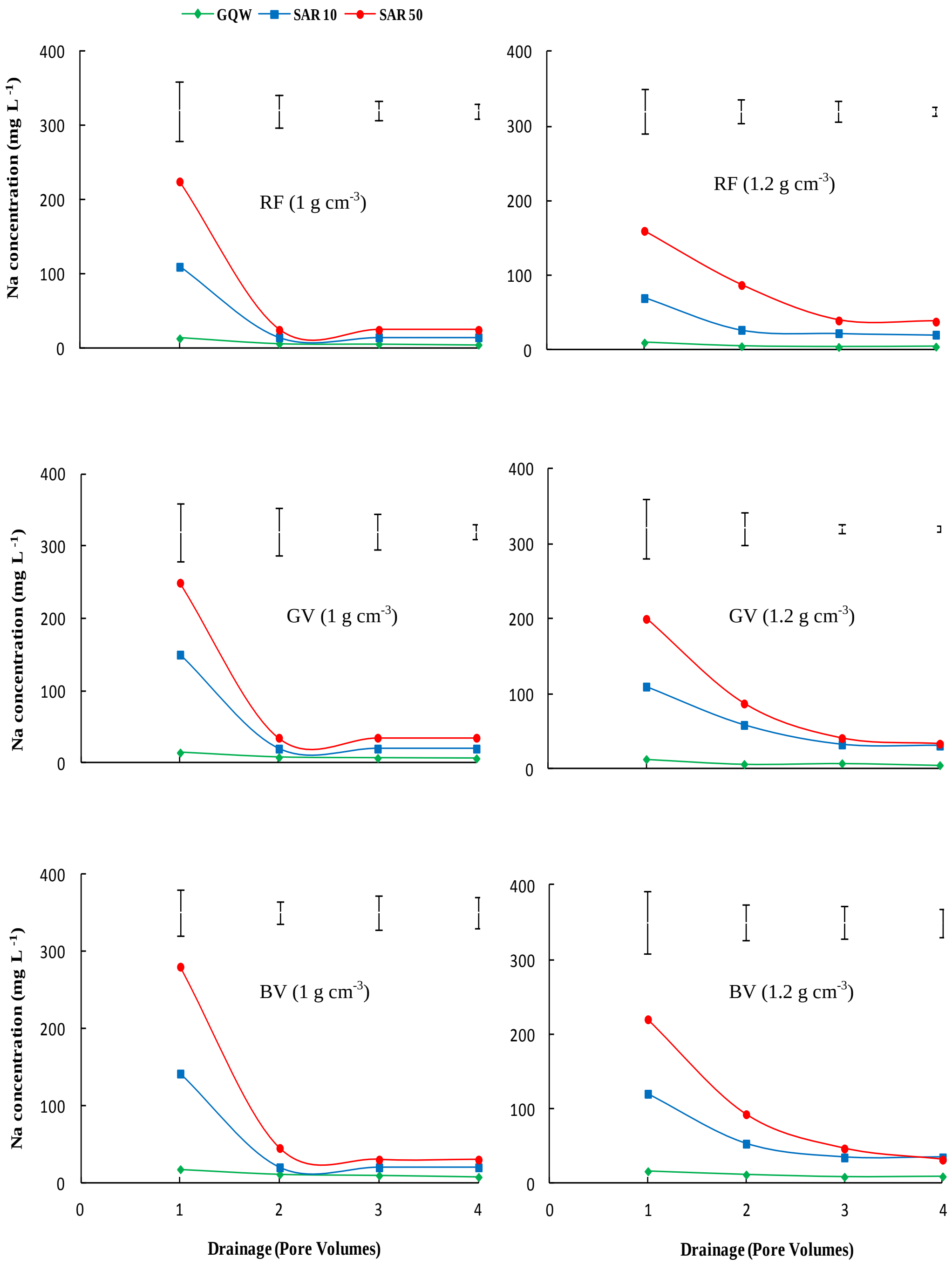


Figure 6-7 Na concentration in the leachate (g L-1) from three clay soils (RF, GV and BV) packed at two densities (1 and 1.2 g cm-3) and treatment with different water qualities (GQW, SAR 10, SAR 50) after applying RW. Bars on the graph are LSD (α = 0.05) between treatments.

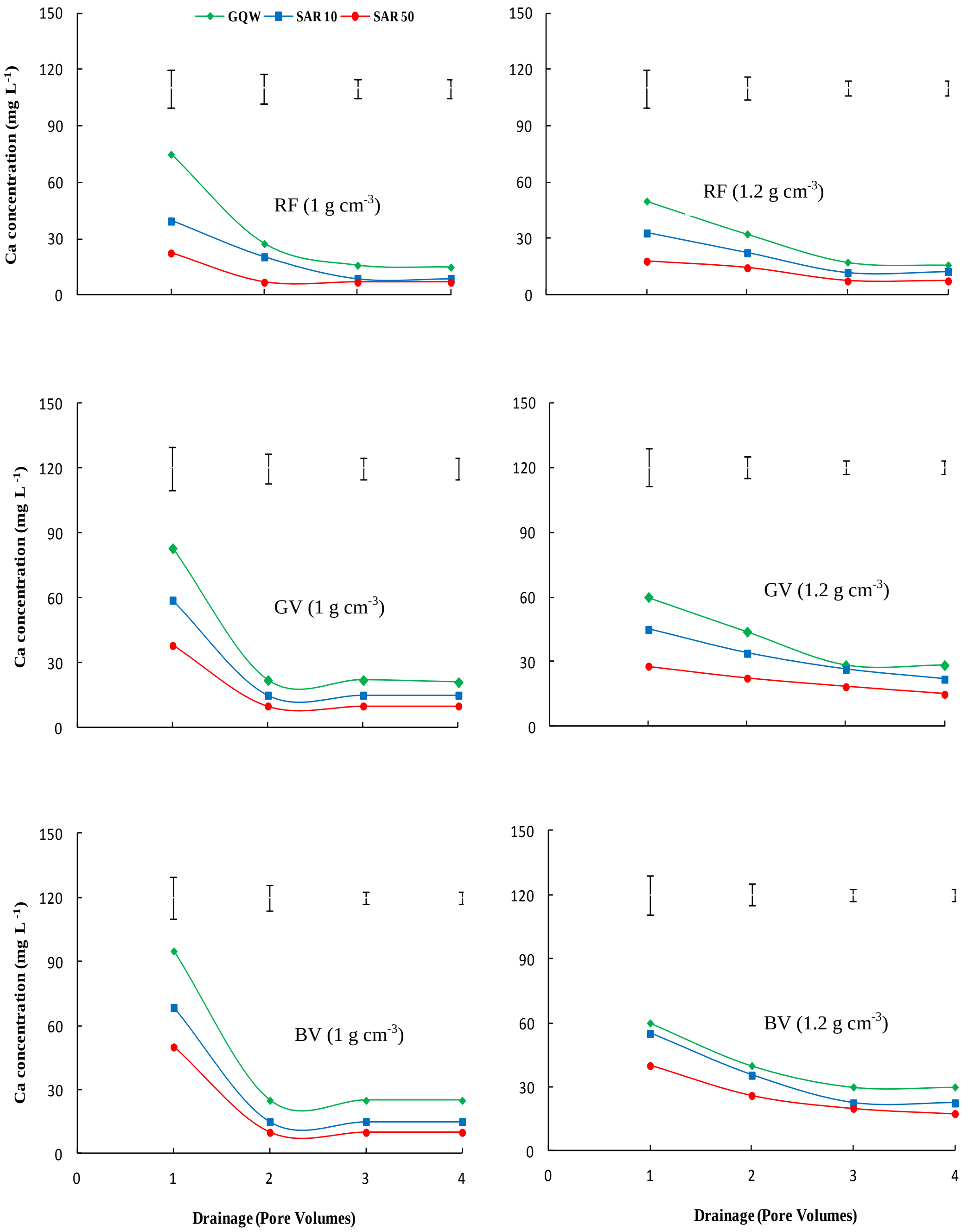


Figure 6-8 Ca concentration in the leachate (g L^{-1}) from three clay soils (RF, GV and BV soils) packed at two densities (1 and 1.2 g cm^{-3}) and treatment with different water qualities (GQW, SAR 10, SAR 50) after applying RW. Bars on the graph are LSD ($\alpha = 0.05$) between treatments.

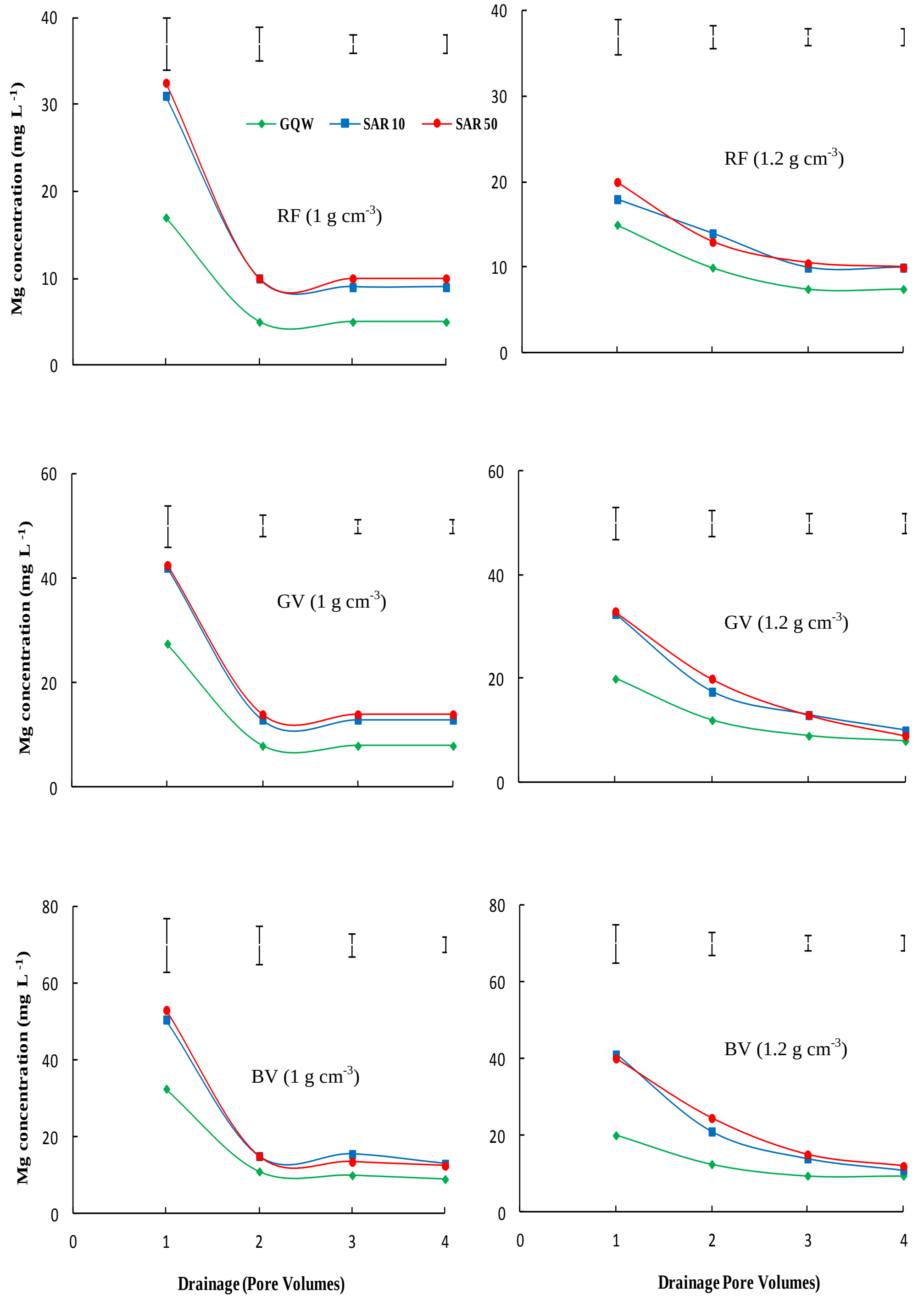


Figure 6-9 Mg^{+2} concentration in the leachate (g L^{-1}) from three clay soils (RF, GV and BV) packed at two densities (1 and 1.2 g cm^{-3}) and treatment with different water qualities (GQW, SAR 10, SAR 50) after applying RW. Bars on the graph are LSD ($\alpha = 0.05$) between treatments.

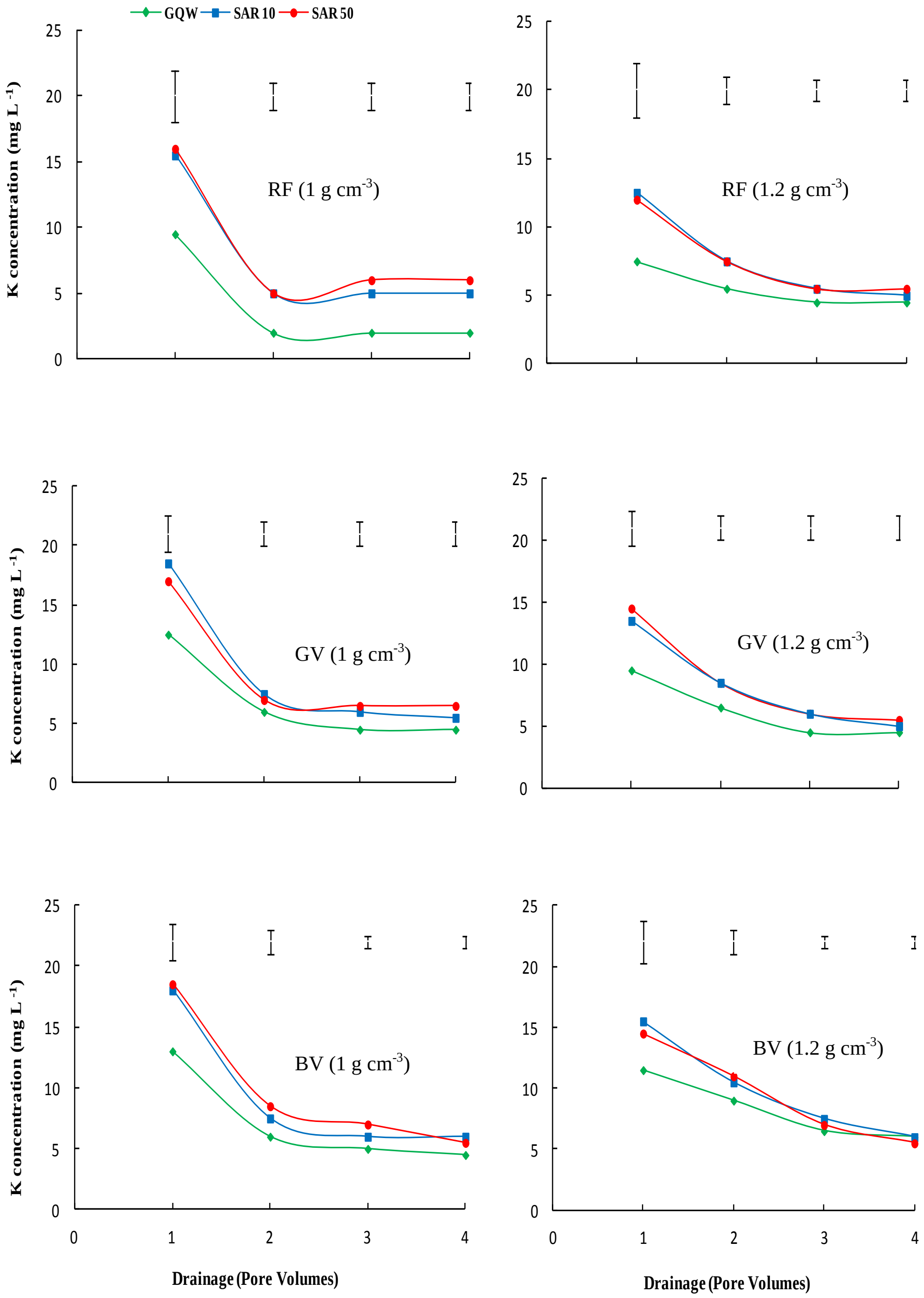


Figure 6-10 K concentration in the leachate (g L-1) from three soils (RF, GV and BV) packed at two densities (1 and 1.2 g cm-3) and treatment with different water qualities (GQW, SAR 10 and SAR 50) after applying RW. Bars on the graph are LSD (α = 0.05) between treatments.

6.3.5 Exchangeable cations and soil ESP

The application of GQW to the soil columns generally resulted in a significant decrease in exchangeable Na^+, but there was no significant difference in exchangeable Mg^{2+} and K^+ throughout the soil columns (Table 6-1). For all soils, applying SAR10 and SAR50 water significantly ($P<0.05$) increased the concentration of exchangeable Na^+ in the surface 1 cm of the soil column compared with deeper (e.g. 3 – 4 and 5 – 6 cm) in the columns (Table 6.1). Overall, exchangeable Na^+ was significantly ($P<0.05$) higher in the lower density (1.0 g cm^{-3}) columns compared with the higher density columns. For instance, exchangeable Na^+ in the RF soil treated with SAR 50 water was 10 and 16 times higher at the soil surface than the initial exchangeable Na^+ concentration for ρ_b = 1 and 1.2 g cm^{-3}, respectively. At depth (5 to 6 cm), exchangeable Na^+ was found to be between 5 and 11 times higher than the initial value of exchangeable Na^+. The changes in exchangeable Na^+ were greater at the soil surface and decreased progressively with depth for all soils. However, the exchangeable Na^+ concentration with depth was significantly ($P<0.05$) influenced by soil density with the lower density soils having higher ESP at shallow depths compared with the higher density soils. For example, the ESP at the soil surface of the BV soil treated with SAR 50 was 29 and 23 for ρ_b = 1 and 1.2 g cm^{-3}, respectively. These values were significantly lower (ESP = 17 and 13) at a depth of 5-6 cm. For the same soil treated with GQW, the ESP at the soil surface was 1 and 2, respectively for ρ_b = 1 and 1.2 g cm^{-3} columns. However, the ESP was 3 in both density treatments at a depth of 5-6 cm.

Table 6-1 Exchangeable cations in the three soils packed at 1 and 1.2 g cm-3 treated with GQW, SAR 10 or SAR 50 after subsequently applying 4 PV of RW

Depth (cm)	ρ_b (g cm^{-3})	Na Cmol (+) kg^{-1}				Ca Cmol (+) kg^{-1}				Mg Cmol (+) kg^{-1}				K Cmol (+) kg^{-1}			
		Initial Ex.Na	**GQW**	**SAR 10**	**SAR 50**	**Initial Ex.Ca**	**GQW**	**SAR 10**	**SAR 50**	**Initial Ex.Mg**	**GQW**	**SAR 10**	**SAR 50**	**Initial Ex.K**	**GQW**	**SAR 10**	**SAR 50**
									RF soil								
0-1 cm	1	0.42^{a}	0.04^{bA}	4.0^{cA}	7.0^{dA}	9.5^{a}	12.3^{bA}	11.4^{cA}	5.1^{dA}	7.1^{a}	4.1^{bA}	2.0^{bA}	$4.0^{d A}$	0.80^{a}	0.70^{bA}	0.4^{cA}	0.5^{cA}
	1.2	0.42^{a}	0.11^{bB}	3.0^{cB}	5.0^{dB}	9.5^{a}	11.3^{bB}	9.5^{cA}	8.9^{dB}	7.1^{a}	6.0^{bB}	4.8^{bB}	4.5^{cB}	0.80^{a}	0.80^{aB}	0.5^{bB}	0.5^{bB}
3-4 cm	1	0.42^{a}	0.12^{bA}	2.0^{cA}	5.0^{dA}	9.5^{a}	12.4^{bcA}	12.3^{cA}	7.8^{dA}	7.1^{a}	5.2^{aA}	3.9^{aA}	4.6^{cA}	0.80^{a}	0.50^{bA}	0.5^{cA}	0.5^{cA}
	1.2	0.42^{a}	0.15^{bB}	2.0^{cB}	3.0^{dB}	9.5^{a}	11.0^{bB}	10.9^{cB}	10.0^{dB}	7.1^{a}	6.7^{abA}	5.0^{abA}	5.0^{bB}	0.80^{a}	0.70^{bB}	0.5^{cB}	0.6^{cB}
5-6 cm	1	0.42^{a}	0.10^{bA}	2.0^{cA}	4.0^{dA}	9.5^{a}	11.7^{bcA}	11.2^{cA}	9.1^{dA}	7.1^{a}	6.0^{bA}	3.9^{bA}	5.0^{cA}	0.80^{a}	0.80^{bA}	0.5^{cA}	0.5^{cA}
	1.2	0.42^{a}	0.17^{bB}	1.9^{cB}	3.0^{dB}	9.5^{a}	10.1^{abB}	11.4^{bB}	10.1^{cA}	7.1^{a}	7.0^{aA}	3.8^{aA}	5.0^{bB}	0.80^{a}	0.70^{bA}	0.6^{cA}	0.6^{cA}
									GV soil								
0-1 cm	1	1.30^{a}	0.14^{bA}	7.0^{cA}	12.0^{dA}	17.2^{a}	24.0^{bA}	20.0^{cA}	13.0^{dA}	15.4^{a}	11.0^{bA}	8.0^{bA}	10.3^{cA}	1.35^{a}	1.3^{bA}	0.8^{cA}	0.6^{cA}
	1.2	1.30^{a}	0.31^{bB}	5.0^{cB}	8.0^{dB}	17.2^{a}	19.8^{bB}	17.9^{dB}	14.9^{dA}	15.4^{a}	14.8^{bB}	11.1^{bB}	11.3^{cB}	1.35^{a}	1.2^{bA}	1.1^{bB}	1.0^{bB}
3-4 cm	1	1.30^{a}	0.73^{bA}	3.5^{cA}	9.0^{dA}	17.2^{a}	21.3^{bA}	20.1^{cA}	15.1^{dA}	15.4^{a}	12.0^{bA}	11.6^{bA}	12.6^{cA}	1.35^{a}	1.2^{bA}	1.1^{bA}	1.0^{bA}
	1.2	1.30^{a}	0.80^{bB}	3.0^{bB}	4.0^{dB}	17.2^{a}	20.3^{bB}	18.1^{aB}	16.5^{cB}	15.4^{a}	15.0^{bB}	14.0^{bB}	14.9^{bB}	1.35^{a}	1.2^{bA}	1.2^{bcA}	0.9^{cB}
5-6 cm	1	1.30^{a}	0.70^{bA}	3.0^{cA}	7.0^{dA}	17.2^{a}	18.1^{aA}	18.9^{bA}	12.9^{dA}	15.4^{a}	16.0^{bA}	14.2^{bA}	14.0^{cA}	1.35^{a}	1.1^{bA}	1.0^{cA}	0.9^{cA}
	1.2	1.30^{a}	1.12^{bB}	3.0^{cB}	4.0^{dB}	17.2^{a}	18.3^{bB}	19.0^{aB}	18.1^{aB}	15.4^{a}	16.0^{aB}	13.6^{aB}	15.3^{bB}	1.35^{a}	1.2^{bA}	1.1^{bcA}	1.1^{cB}
									BV soil								
0-1 cm	1	1.45^{a}	0.50^{bA}	8.0^{cA}	14.0^{dA}	21.9^{a}	33.0^{bA}	29.0^{bA}	18.0^{cA}	20.2^{a}	11.8^{bA}	7.0^{bA}	11.9^{cA}	2.94^{a}	2.5^{bA}	1.9^{cA}	2.2^{cA}
	1.2	1.45^{a}	0.90^{bB}	7.1^{cB}	11.0^{dB}	21.9^{a}	27.2^{bB}	25.5^{bB}	17.5^{cA}	20.2^{a}	17.1^{bB}	13.1^{bB}	16.1^{dB}	2.94^{a}	2.5^{bB}	2.3^{cB}	2.4^{cB}
3-4 cm	1	1.45^{a}	0.88^{bA}	4.7^{cA}	11.0^{dA}	21.9^{a}	32.1^{bA}	27.1^{bA}	19.2^{cA}	20.2^{a}	13.3^{bA}	14.5^{bA}	15.8^{cA}	2.94^{a}	2.6^{bA}	2.4^{cA}	2.3^{cA}
	1.2	1.45^{a}	1.0^{bB}	3.0^{cB}	5.0^{dB}	21.9^{a}	26.0^{bA}	24.1^{cB}	22.0^{dB}	20.2^{a}	20.0^{aB}	16.8^{aB}	19.2^{bB}	2.94^{a}	2.6^{bB}	2.4^{cB}	2.6^{cB}
5-6 cm	1	1.45^{a}	1.28^{bA}	4.0^{cA}	$8 .0^{dA}$	21.9^{a}	30.3^{bA}	26.0^{bA}	20.0^{cA}	20.2^{a}	14.3^{bA}	16.0^{bA}	18.0^{cA}	2.94^{a}	2.7^{bA}	2.6^{cA}	2.4^{cA}
	1.2	1.45^{a}	1.50^{bB}	4.0^{cB}	6.0^{dB}	21.9^{a}	24.1^{bB}	22.0^{cB}	22.1^{cA}	20.2^{a}	20.0^{aA}	18.1^{aA}	17.3^{bB}	2.94^{a}	2.9^{bB}	2.7^{cb}	2.8^{cB}

Use lower case letters for comparisons between treatments and upper case letters for comparisons between within treatments.

Table 6-2 ESP of the RF, GV and BV soils treated with GQW, SAR 10 or SAR 50 at three depths after applying RW

Depth (cm)	ρ_b (g cm^{-3})	GQW	SAR 10	SAR 50
		RF soil		
0-1 cm	1	0.2^{aA}	21.0^{bA}	36.8^{cA}
	1.2	0.5^{aB}	16.3^{bB}	26.3^{cB}
3-4 cm	1	0.6^{aA}	10.5^{bA}	26.3^{cA}
	1.2	0.7^{aB}	10.5^{bB}	15.8^{cB}
5-6 cm	1	0.5^{aA}	10.9^{bA}	21.0^{cA}
	1.2	0.8^{aB}	10.5^{bB}	15.7^{cB}
		GV soil		
0-1 cm	1	0.3^{aA}	19.2^{bA}	32.9^{cA}
	1.2	0.8^{aB}	13.7^{bB}	21.9^{cB}
3-4 cm	1	2.0^{aA}	9.6^{bA}	24.6^{cA}
	1.2	2.9^{aA}	8.2^{bB}	10.9^{cB}
5-6 cm	1	1.9^{aA}	8.2^{bA}	19.2^{cA}
	1.2	3.0^{aB}	8.2^{bB}	10.9^{cB}
		BV soil		
0-1 cm	1	1.0^{aA}	16.6^{bA}	29.1^{cA}
	1.2	1.8^{aB}	14.7^{bB}	22.9^{cB}
3-4 cm	1	1.83^{aA}	9.3^{bA}	22.9^{cA}
	1.2	2.0^{aB}	6.2^{bB}	10.4^{cB}
5-6 cm	1	2.6^{aA}	8.3^{bA}	16.6^{cA}
	1.2	3.1^{aB}	8.3^{bB}	12.5^{cB}

6.3.6 Changes in soil porosity

The analyses of soil images from different depths in the columns (Figure 6-11 to 6-14) show that water quality and soil density had a significant effect on total areal soil porosity. The influence of these parameters on soil areal porosity was observed on the soil surface as well as at depth. For the RF soil treated with GQW water, the total areal porosity was 60% and 40% for the low and high density soils, respectively (Figure 6-15). Total areal porosity in GQW columns did not change significantly ($P>0.05$) with depth. However, application of SAR 50 water reduced soil porosity by up to 43% and 30% (compared with GQW treated soil) for $\rho_b = 1$ and 1.2 g cm^{-3} (Figure 6.15).

For the swelling BV soil treated with GQW the total areal porosity at all depths were approximately 28% and 18% for $\rho_b = 1$ and 1.2 g cm^{-3}, respectively. The application of SAR 50 water to BV soil reduced total areal porosity significantly by up to 50% and 44% for $\rho_b = 1$ g cm^{-3} and 1.2 g cm^{-3}, respectively (Figure 6.15).

The reduction in soil areal porosity observed after the application of RW was greater in soils that had been treated with saline-sodic water. For the RF soil post-treatment with GQW, the areal porosity at the soil surface were 53 and 30 % for $\rho_b = 1$ and 1.2 g cm^{-3}, respectively. When RW was applied to $\rho_b = 1$ g cm^{-3} RF soil columns after treatment with SAR 50 water, the areal porosity at the surface was > 50% but declined significantly to 30% at depths of 4 and 6 cm (Figure 6-16). However, for the same soil packed at 1.2 g cm^{-3}, the areal porosity at the soil surface was only 8 % while the areal porosity at 6 cm depth was significantly larger at 20% (Figure 6.16).

For the BV, applying the RW after any of the water quality treatment significantly reduced the areal porosity throughout the column (Figure 6-15 and 6-16). However, the reductions in porosity were larger in the surface layer than at depth in the SAR 10 and 50 treatments (Figure 6-16). There was no significant difference in the areal porosity at 6 cm depth between the treatments for the 1 g cm^{-3} columns and 2 – 6 cm depth for the 1.2 g cm^{-3} columns.

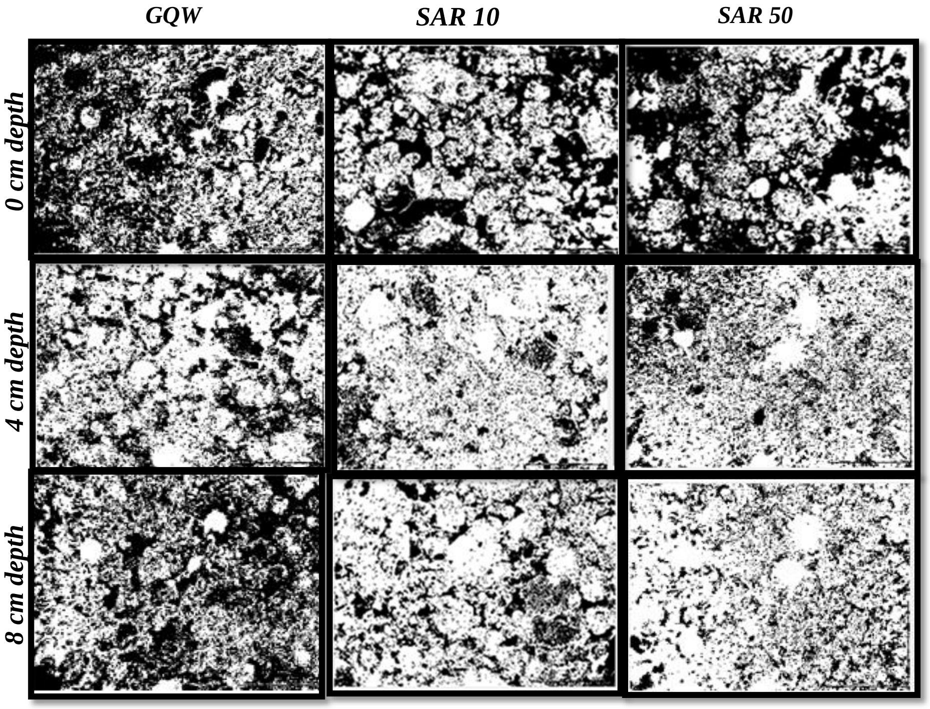


Figure 6-11 Selected images of surface soil blocks of RF soil packed at (1 g cm-3) treated with GQW, SAR 10 and SAR 50 at different depths (0, 4 and 8 cm).

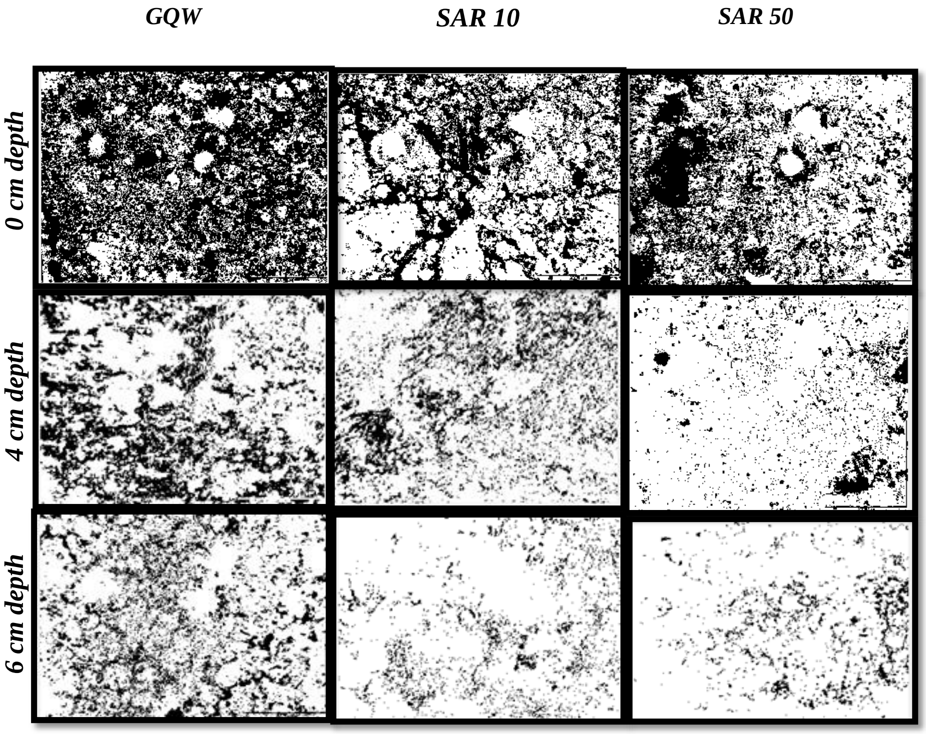


Figure 6-12 Selected images of surface soil blocks of RF soil packed at (1 g cm-3) treated with GQW, SAR 10 and SAR 50 at different depths (0, 4 and 6 cm) after applying RW.

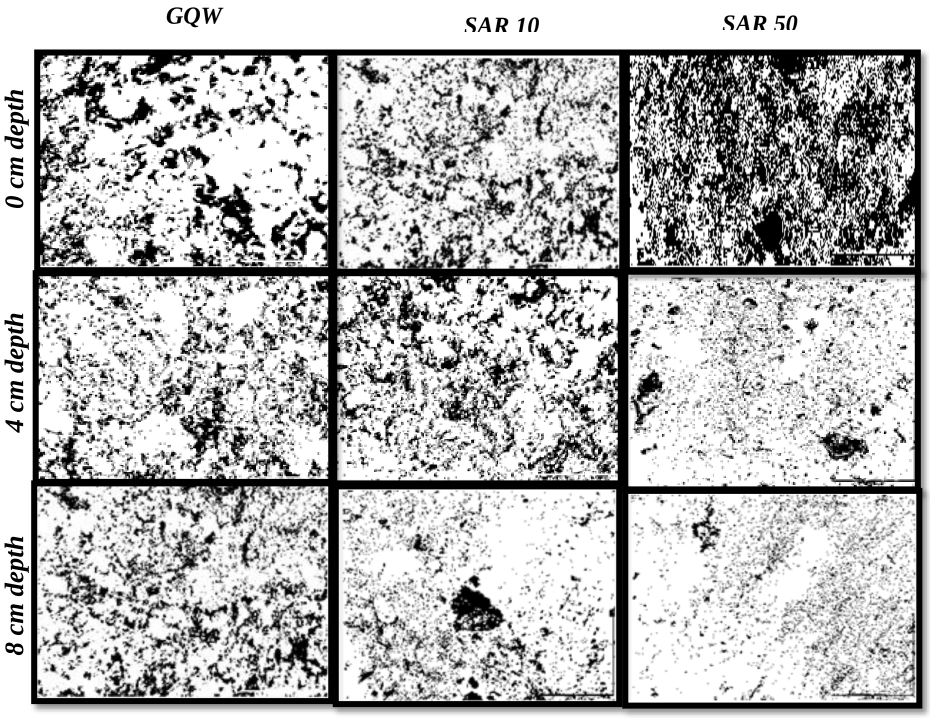


Figure 6-13 Selected images of surface soil blocks of BV soil packed at (1 g cm^{-3}) treated with GQW, SAR 10 and SAR 50 at different depths (0, 4 and 8 cm).

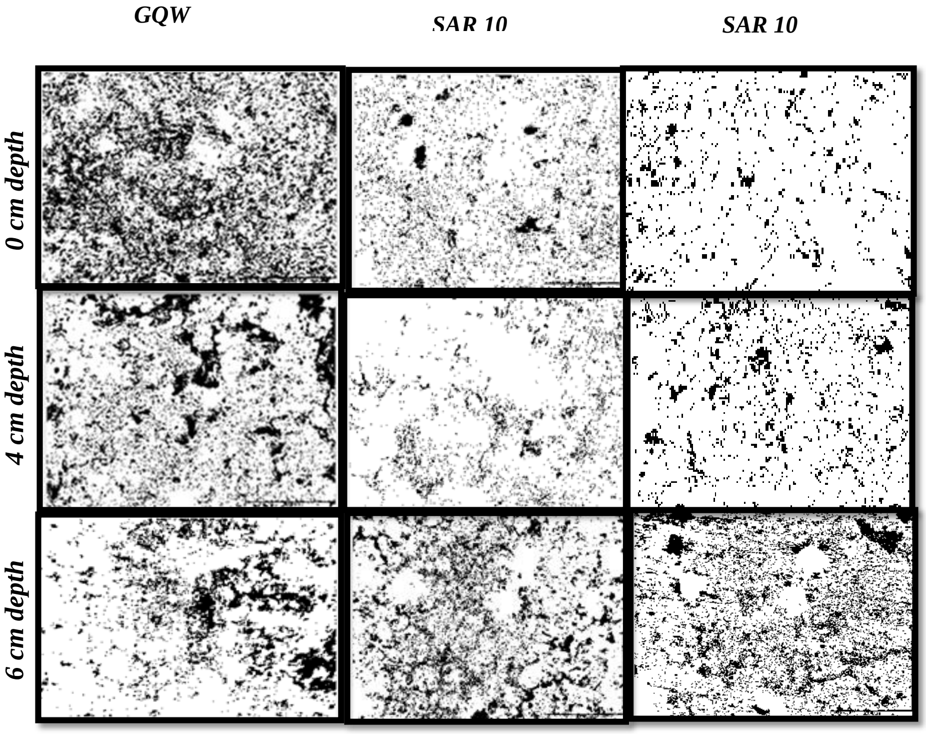


Figure 6-14 Selected images of surface soil blocks of BV soil packed at (1 g cm-3) treated with GQW, SAR 10 and SAR 50 at different depths (0, 4 and 6 cm) after applying RW.

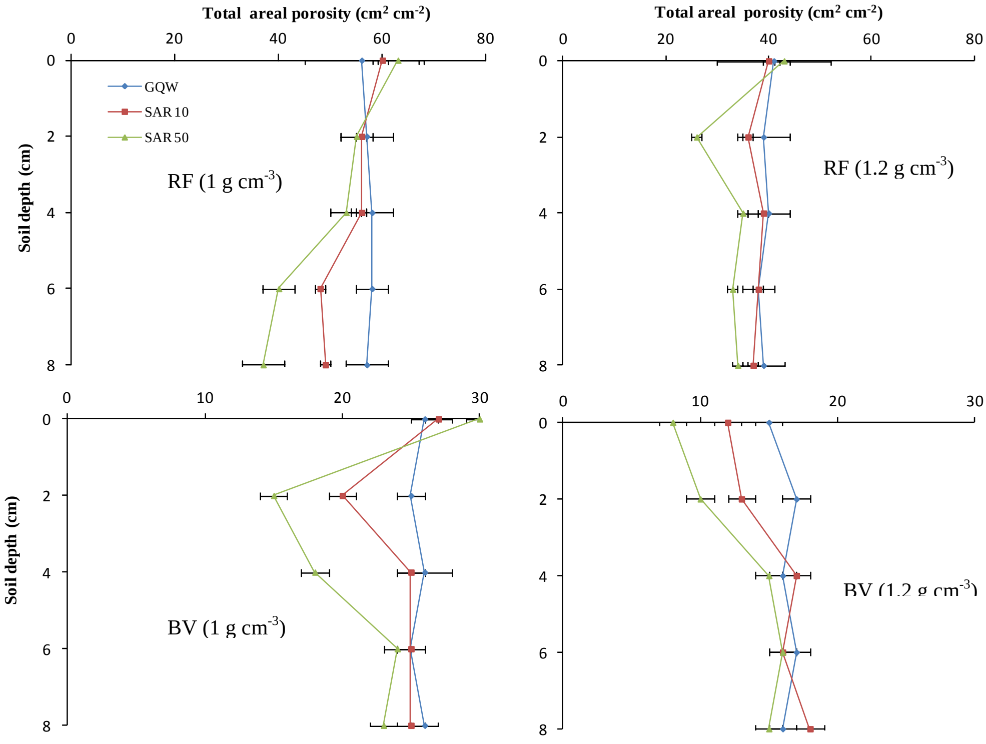


Figure 6-5 Total areal porosity distribution for RF and BV soil columns packed at 1and 1.2 g cm-3 treated with GQW, SAR 10 or SAR 50 water.

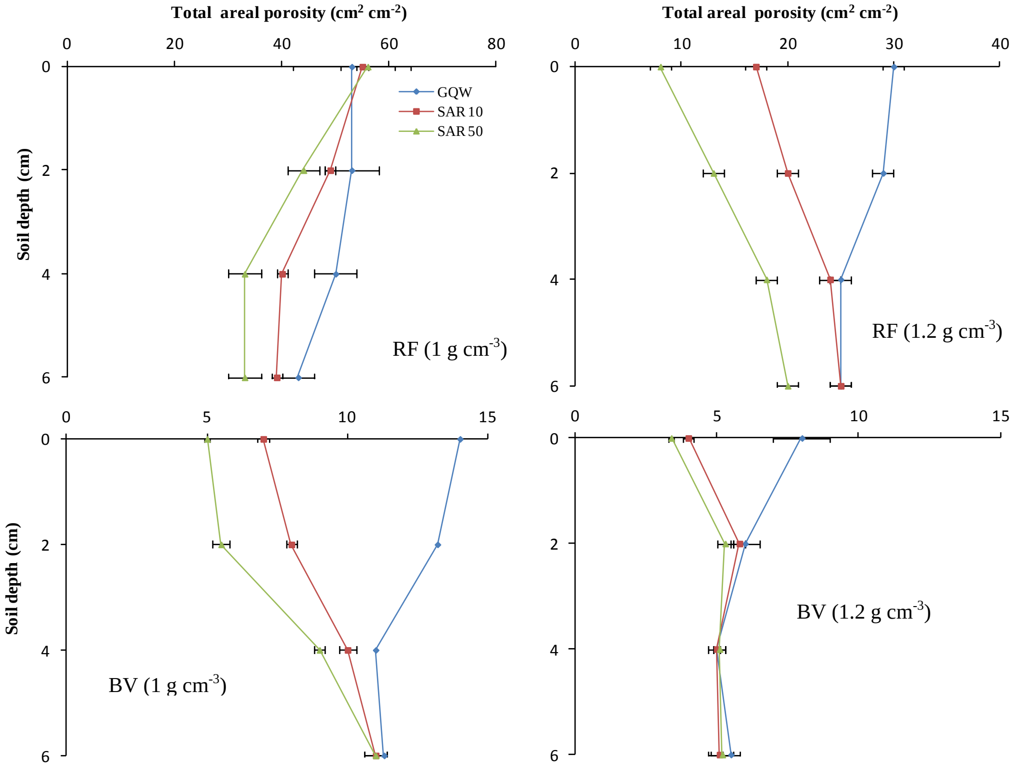


Figure 6-16 Total areal porosity distribution after applying RW to RF and BV soil columns packed at 1 or 1.2 g cm-3 and treated with GQW, SAR 10 or SAR 50 water.

6.3.7 Pore size distribution

Soil macropores were significantly affected by packing density and the SAR of the solution applied (Figure 6-17 and 6-18). For example, in the RF soil packed at 1 g cm^{-3} and treated with GQW macropores represented approximately 55% of the total porosity throughout the soil columns (Figure 6-17). Increasing the packing density to 1.2 g cm^{-3} decreased the macropores to approximately 25% of total porosity. Application of SAR 50 water to the RF soil packed at 1 g cm^{-3}, reduced macro-porosity to about 35% at 8 cm depth and when packed at 1.2 g cm^{-3} reduced macropores significantly to approximately 15% throughout the columns (Figure 6-17).

Similar results were found for the BV soil (Figure 6-18). However, while macropores were the dominant size in the RF soil packed at 1 g cm^{-3} (Figure 6-17) micropores were the dominant pores for all BV soil treatments (Figure 6-18). For the higher packing density treatment, there were also significant differences in the micropores at the surface (approximately 80%) and at 4 – 8 cm depth (approximately 65%) in the SAR 10 and 50 treatments (Figure 6-18).

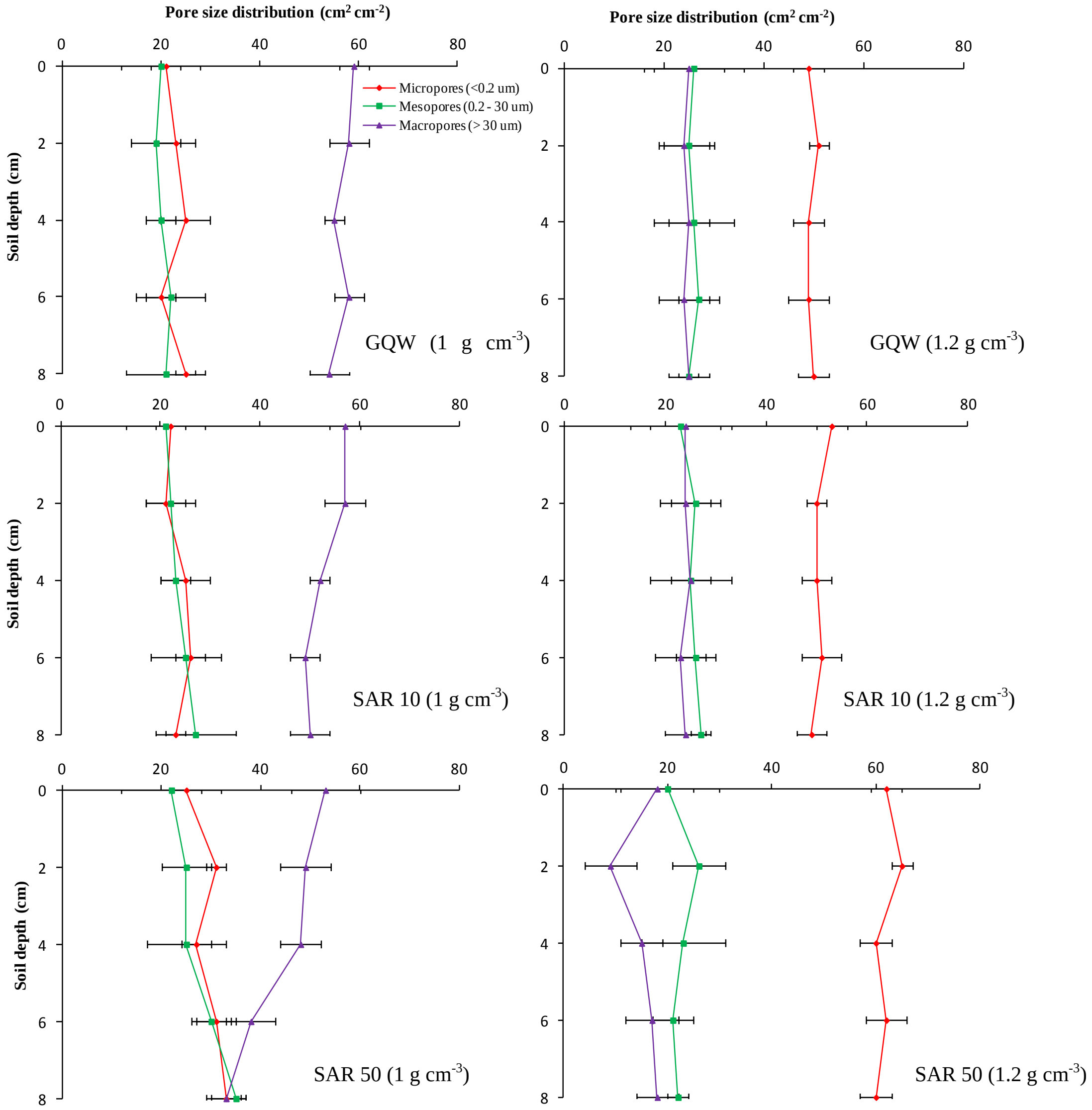


Figure 6-17 Pore size distribution for RF soil packed at 1 or 1.2 g cm-3 and treated with GQW, SAR 10 and SAR 50 water.

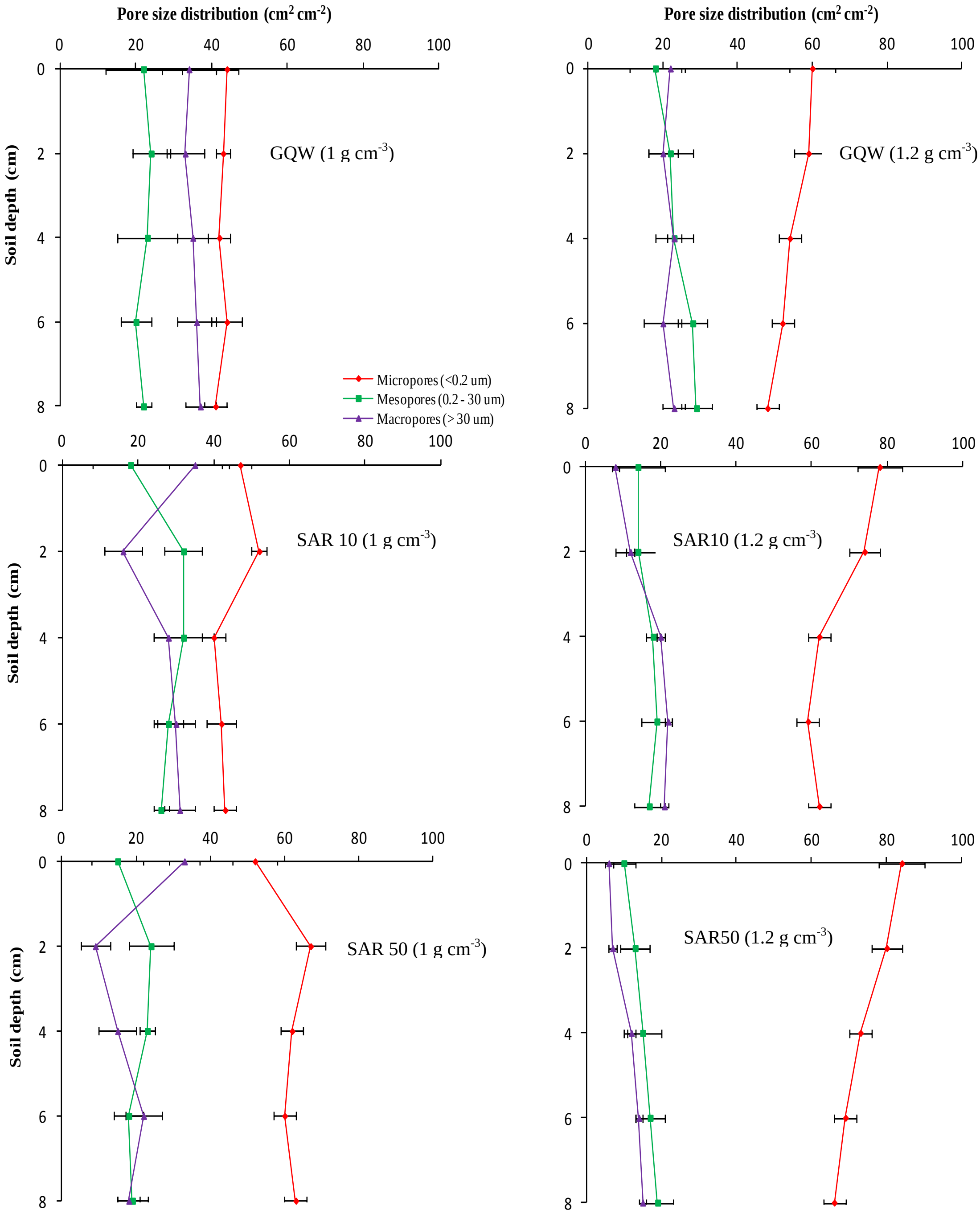


Figure 6-18 Pore size distribution for BV soil packed at 1 or 1.2 g cm-3 and treated with GQW, SAR 10 and SAR 50 water.

Applying RW to the columns after either GQW or saline-sodic treatment generally significantly reduced the proportion of macro and mesopores and increased the proportion of micropores (compare Figure 6-17 and 6-18 with 6-19 and 6-20). Pore size distributions within the soil columns were also affected by soil packing density and the SAR of the previous water treatment (Figure 6-19 and 6-20). For instance, when RW was applied to RF soil packed at 1 g cm^{-3} and after treatment with SAR 50 water (Figure 6-19), the macropores decreased from 33% at the surface to 13% at a depth of 8 cm. This reduction in macropores was associated with an equivalent increase in the micropores from 47% at the surface to 74% at 8 cm depth. For the same soil and SAR treatment macropores increased from 3% at the soil surface to 13% at 8 cm depth while the mesopores decreased from 86% at the surface to 63% at 8 cm depth (Figure 6-19).

The reduction in macro and mesopores associated with applying the RW was more severe for the BV soil (Figure 6-20) compared to the RF soil (Figure 6-19). For example, > 80% of the total porosity at a depth of 4 – 8 cm was micropores in all BV treatment columns. Similarly, except for the 1 g cm^{-3} GQW treatment (73% micropores), the surface layer in all treatments was ≥ 90% micropores.

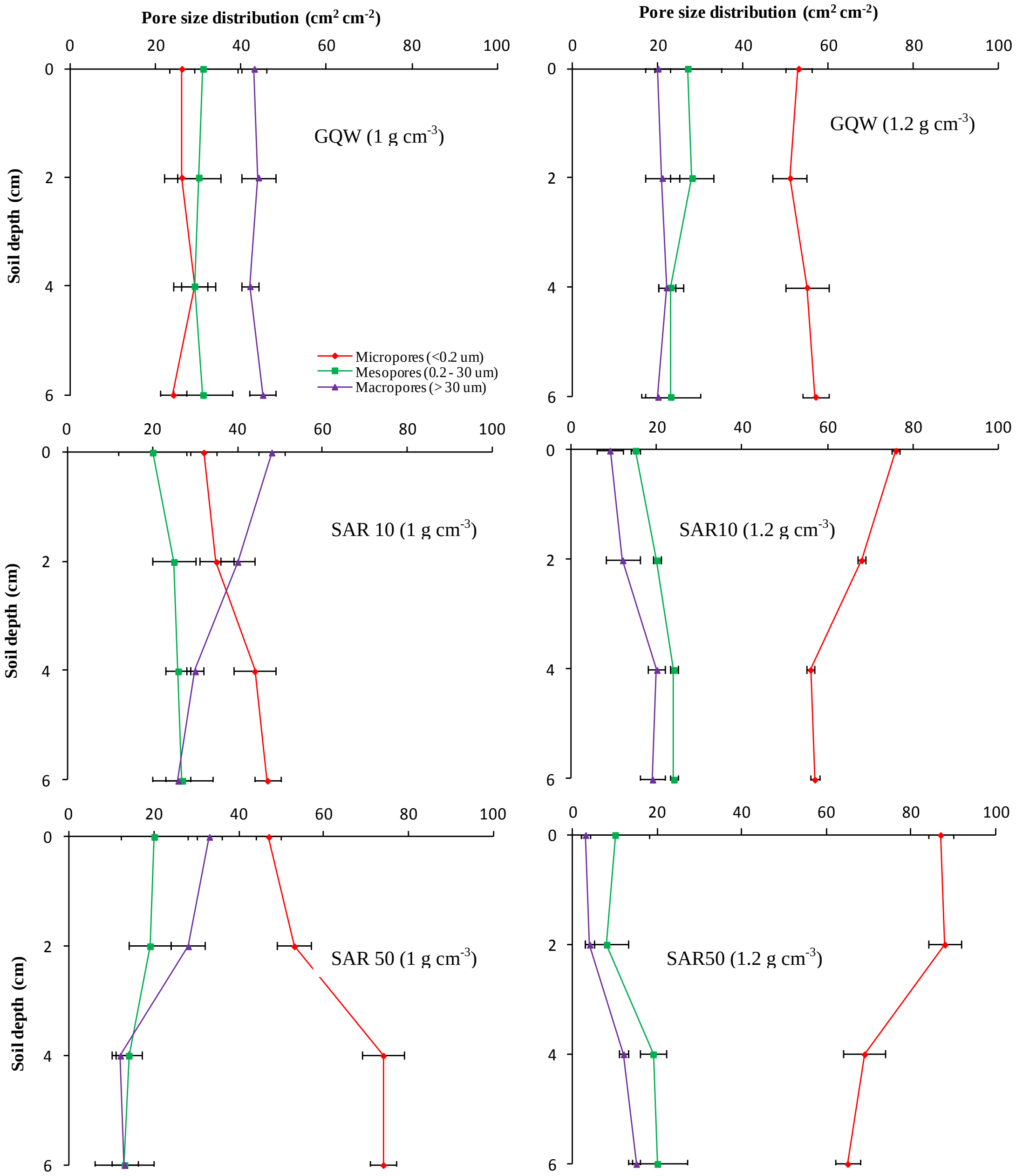


Figure 6-19 Pore size distribution for RF soil packed at 1 or 1.2 g cm-3 and treated with GQW, SAR 10 or SAR 50 water after applying RW.

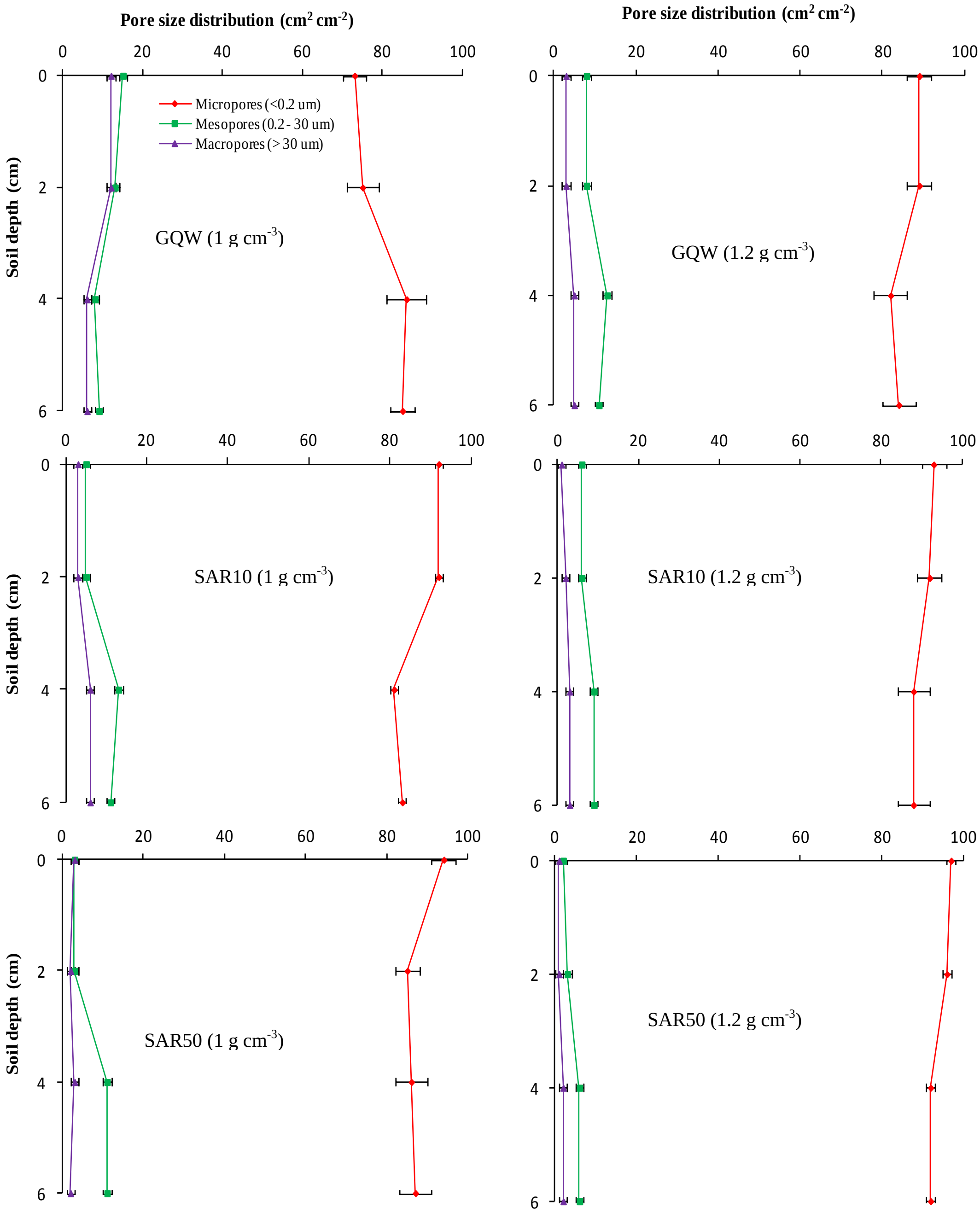


Figure 6-20 Pore size distribution for BV soil packed at 1 or 1.2 g cm-3 and treated with GQW, SAR 10 or SAR 50 water after applying RW

6.4 Discussion

6.4.1 Effect of soil density and applied water quality on saturated hydraulic conductivity

Generally K_s decreased for all soils when RW was applied to the columns after application of the GQW or saline-sodic water. However, the reductions were more pronounced with RW application after application of the more saline-sodic water.

The EC of the drainage leachate is generally considered to be similar to the EC of the soil solution (e.g. MacDonald et al 2004; Corwin and Lesch, 2005). In the early stages of leaching with RW (1 PV) the leachate EC was generally > 0.5 dS m^{-1} maintaining flocculation and allowing soils to remain permeable (Hanson et al. 1999). As EC decreased with further leaching of soluble ions (> 2 PV), K_s decreased significantly (Figure 6-4). This may be due to increased osmotic potential between the bulk soil solution and the interior of soil aggregates resulting in water flow into the micropores. This internal process in the micropores causes clay swelling and dispersion consistent with the dispersed clay noted in the leachate.

Clay dispersion and movement resulted in reductions in the total porosity and partial blockage of macropores as shown by the reduction in macropores (Figure 6-19 and 6-20). The reduction in both total porosity and proportion of macropores have significantly reduced water movement under saturated conditions and reduced K_s (Ayers and Westcot, 1976; Hanson et al. 1999). Soil pore blockage occurred in different soil layers within the column depending on packing density and soil type (Figure 6-15 and 6-16).

K_s will be a function of the total porosity and macroporosity of the most restriction layer. The K_s during leaching with RW in low density columns were generally higher and reached steady state with less drainage compared to compacted columns possibly due to fast leaching of soluble salts within macropores. In contrast, the higher proportion of micropores in the compacted columns led to slow leaching of soluble salts resulting in slower reductions in K_s with drainage.

The reduction in K_s with application of RW was significantly affected by the increased soil ESP that was produced by the application of saline-sodic water. The reductions in K_s were more pronounced in soil treated with SAR 50 compared to SAR 10 and GQW treatments. Dispersed clay concentrations in the leachate were also generally higher with increasing soil ESP (Figure 6-6). This resulted from the adsorption of Na^+ cations (with relatively large hydrated radius and single charge) onto the clay exchange surface causing physical separation of the clay particles and consequently, clay dispersion and reduced K_s through soil blockage (Buckman and Brady, 1967; Chen and Banin, 1975; van de Graaff and Paterson, 2001).

The BV and GV soils generally had a lower K_s compared to the RF soil. These differences are most likely related to clay type as the RF soil is dominated by kaolinite and has high concentrations of iron oxides and hydroxides. The kaolinite does not swell while the oxides assist in maintaining clay flocculation and reducing dispersion (Deshpande et al., 1968; Goldberg et al., 1988). In comparison, the GV and BV soils that had a higher content of 2:1 clay minerals that may disperse as they become wet (McNeal and Colemam, 1966; Yaron and Thomas, 1968; Ben-Hur et al. 1998).

6.4.2 Factors affecting clay dispersion and soil pore blockage

The reductions in soil K_s during leaching with RW (Figure 6-4) were larger in the low density columns possibly due to rapid reductions in soil EC, particularly in soil with high ESP (e.g. previously treated with SAR 50 water). Low density soil columns maintained significantly higher K_s compared to compacted soils due to the high initial porosity and macropores that enabled dispersed clay to move through the soil column and out in the leachate. When the dispersed clay was trapped within the column, the depth of clay accumulation was affected by soil type, packing density and water quality (Figure 6-17 and 6-20).

The concentration of dispersed clay particles in the leachate was significantly influenced by the packing density of the columns. The concentration of clay particles in the leachate was higher in the soils packed at 1 g cm^{-3} compared with those packed at 1.2 g cm^{-3} (Figure 6-6). The lower leachate clay concentration observed for the high density soils was due to more dispersed clay particles accumulating in the flow path within the soil column producing reduction in the pore size within the zone of accumulation.

It was noted (Figure 6-19) that accumulation tended to occur in the surface layers (0 – 2 cm) if the proportion of micropores was high (e.g. ≥ 50%). However, where the proportion of micropores was small (e.g. < 30%) the development of an accumulation zone depended on the dispersed clay concentration (Figure 6-19). When the clay concentration was small (e.g. GQW applied to RF soil) there was no accumulation within the soil column and the dispersed clay was passed out of the columns. However, when the dispersed clay concentration increased (e.g. RW applied to RF soil previously treated with SAR 10 and SAR 50 water) then the zone of accumulation was 4 – 6 cm below the surface.

The rate of clay migration out of the BV soil columns was close to zero after 2 PV of drainage (Figure 6-6). This may be due to the swelling property of BV soil that led to a narrowing of the soil pores that could be readily blocked and accumulation of dispersed clay in the soil pores (Pupisky and Shain, 1979). Similarly where RW was applied to BV soil treated with SAR 10 water a higher rate of clay discharge was observed after 3 PVs of drainage than from the SAR 50 treatment (Figure 6-6). This may be due to greater clay dispersion in the SAR 50 treatment that caused rapid pore blockage while the smaller amount of clay dispersion in the SAR 10 treatment allowed the soil pores to remain clear and maintain movement of the dispersed clay out of the column.

Increasing soil packing density significantly reduced areal porosity and the proportion of macropores (Figure 6-12). This was due to soil compaction compressing the macropores leading to a decrease in porosity and creating a higher proportion of soil mesopores and micropores (Richard et al. 2001).

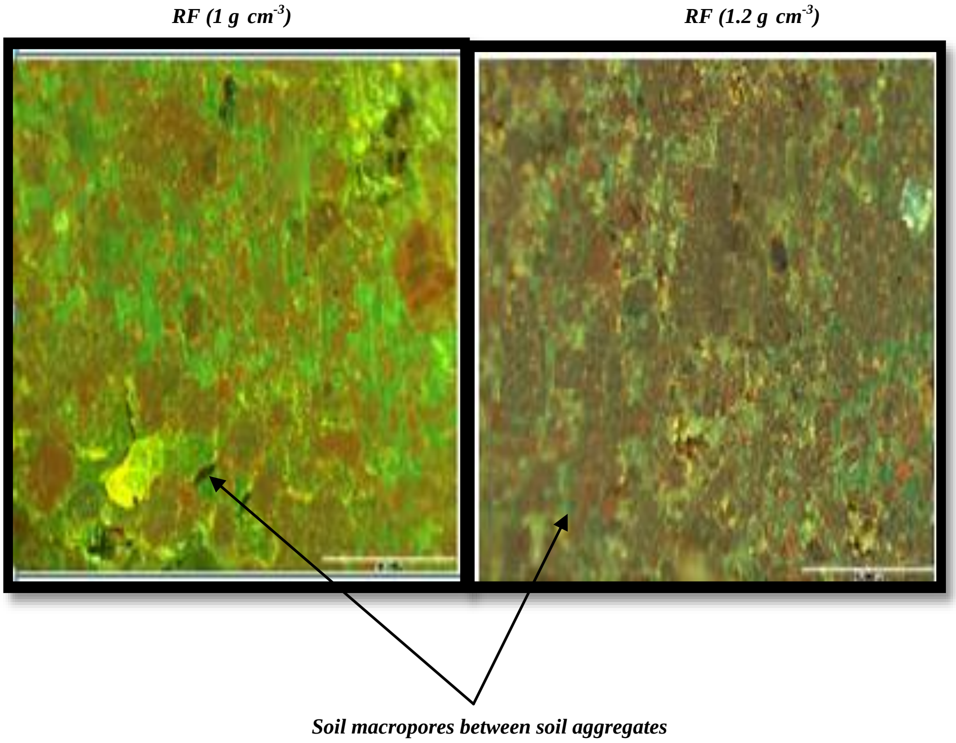


Figure 6-12 Effect of soil compaction on soil macropores between soil aggregates where green and yellow portions represent filled soil pores and red/brown portions represent soil particles.

There was no significant difference in total areal porosity and macroporosity throughout the RF soil column treated with GQW suggesting that GQW had no adverse effect on soil structure. However, application of SAR 50 water resulted in a significant reduction in areal porosity and macropores at a depth of 6 cm for $\rho_b = 1$ g cm^{-3} columns and at a depth of 2 cm for $\rho_b = 1.2$ g cm^{-3} columns. Clay dispersion most likely occurred throughout the column but mainly near the soil surface due to greater leaching and ion exchange as confirmed by the higher ESP (Table. 6-2). The low density soil had more macropores which allowed dispersed clay particles to move downwards. As the dispersed clay concentration increased with movement through the column a critical concentration is reached causing pore blockage within the column at 6 cm depth. In compacted soil, the smaller porosity and pore size led to more rapid blockage of the pores reducing clay movement and producing accumulation at 2 cm depth (Figure 6-22). The significant reduction in macroporosity was mainly due to incomplete blockage of these pores by dispersed clay. This process generated pores with smaller size and consequently increased the proportion of micro and mesopores (Figures 6-23 and 6-24).

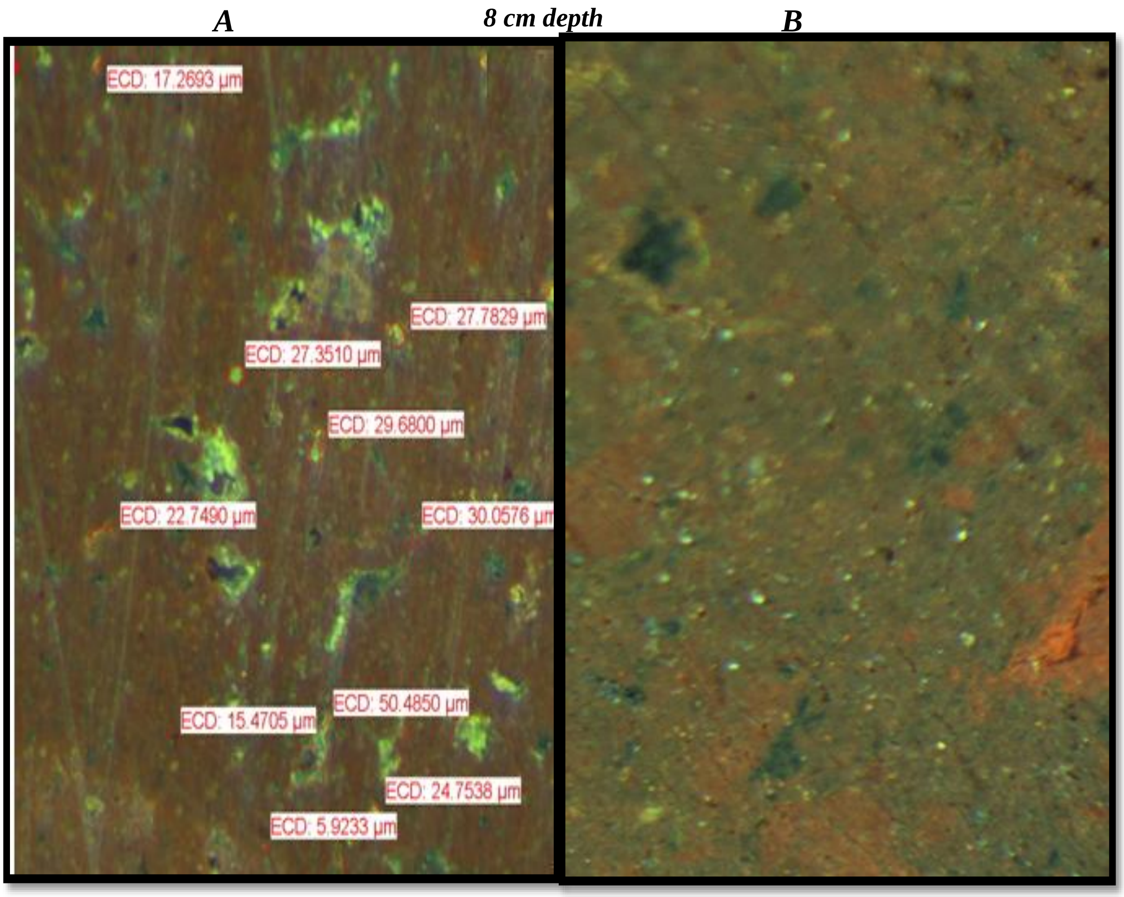
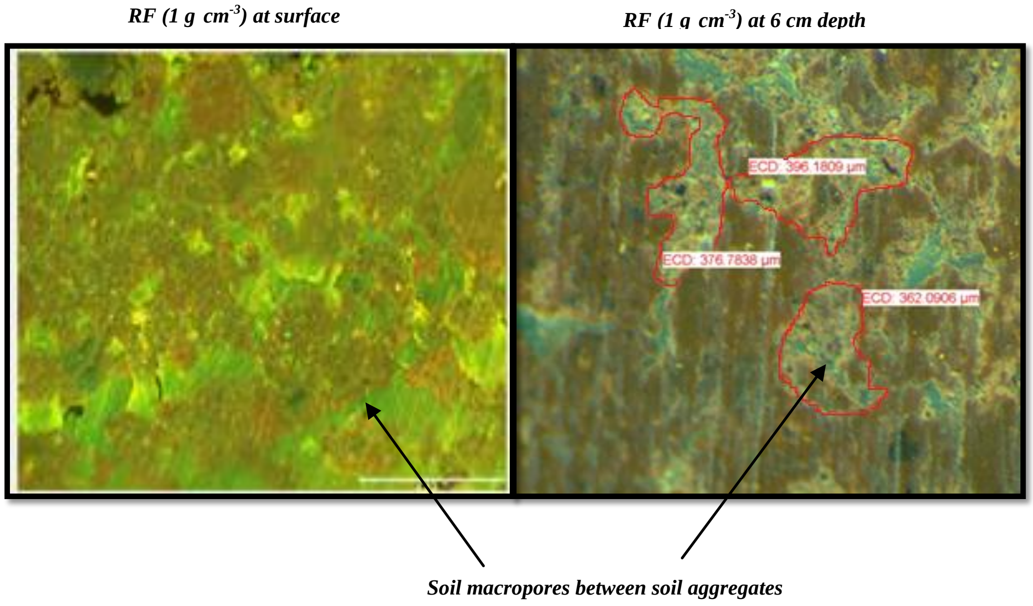


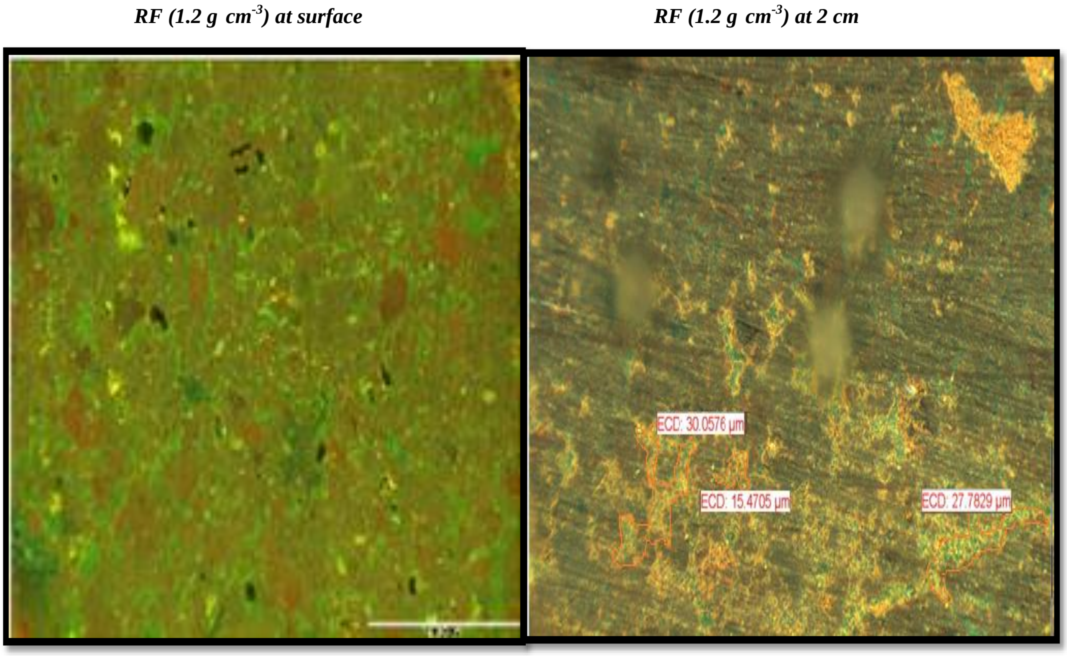
Figure 6-13 Soil pore blockage within RF soil (1 g cm-3) (A) after applying GQW and (B) treated with SAR 50 water where green colours represent soil pores and red/brown colours represent soil particles.


RF (1 g cm^{-3}) at surface

RF (1 g cm^{-3}) at 6 cm depth

Soil macropores between soil aggregates

Figure 6-14 Incomplete blockage of soil macropores for RF soil packed at 1 g cm-3 and treated with SAR 50 water at soil surface and at 6 cm depth where green represents soil pores and red/brown colours represents soil particles.


RF (1.2 g cm^{-3}) at surface

RF (1.2 g cm^{-3}) at 2 cm

Figure 6-15 Incomplete blockage of soil macropores for RF soil packed at 1.2 g cm-3 and treated with SAR 50 water at soil surface and at 2 cm depth where green represents soil pores and red/brown colours represents soil particles.

There was a significant reduction in areal porosity and the proportion of macropores when GQW was applied to the BV soil (Figure 6-25). As there was negligible clay dispersion (Figure 6-6), this difference was attributed to clay swelling (Figure 6-25). This is supported by the observation that porosity and proportion of macropores were similar throughout the soil column (Figure 6-18). However, for the BV soil SAR 50 treatment, dispersed clay movement reduced areal porosity and macroporosity at 2 cm depth in the low density columns and at the surface for the compacted columns (Figure 6-18 and 6-20). This suggested that the BV soil had a reduced ability to migrate dispersed clay through the soil column as a result of swelling impacting on the soil pore size and connectivity. Swelling narrows soil pores and this prevented dispersed clay from migration to a deeper depth. It may also be related to a higher amount of dispersed clay in BV soil due to higher clay content and the sensitivity of the dominant clay (montorillonite) to sodicity.

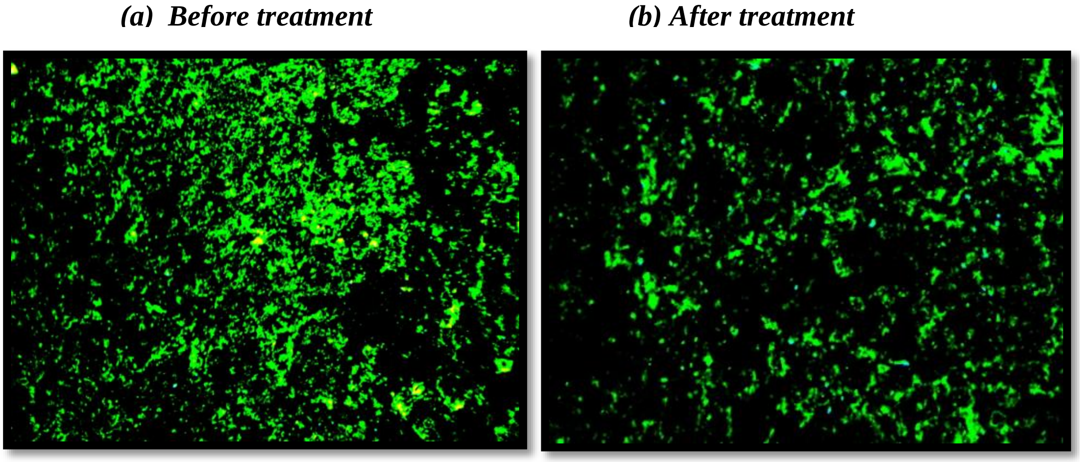

(a) Before treatment *(b) After treatment*

Figure 6-16 Effect of swelling on soil pore-network of BV soil packed at 1 g cm-3, (a) before treatment and (b) after treatment with GQW solution where green represents soil pores and black colours represents soil particles.

The application of RW to the soil columns after treatment with saline-sodic water caused a more severe reduction in areal porosity and macroporosity. These reductions were related to a combined effect of high ESP and reduced EC in the soil solution as soluble salts leached out of the soil profile. Leaching the saline-sodic soils with RW resulted in a decrease in ionic concentrations (Na^+, Ca^{2+}, Mg^{2+} and K^+) of the soil solution and where coupled with the high ESP resulted in an expansion of the diffuse double layer leading to clay dispersion (Naidu and Rengasamy, 1993; Tanji, 1990; Qadir and Schubert, 2002). Hence, degradation of saline-sodic soil aggregates is expected to be more severe at the soil surface due to the more rapid decrease in EC and the high ESP at the surface. Under field conditions the impact of entrapped air release and the stirring action caused by water application will also increase dispersion of soil aggregates at the soil surface (Shainberg and Letey, 1984; Shainberg et al. 1992; Oster and Jayawardane, 1998).
Images of the soil surface after applying RW show almost complete breakdown of the soil aggregates and soil structure (Figure 6-26). Aggregate breakdown might have resulted from osmotic potential differences between the bulk soil solution and the interior of the soil aggregates leading to water flow into the micropores causing clay swelling and dispersion (Ayers and Westcot, 1976; Hanson et al. 1999).

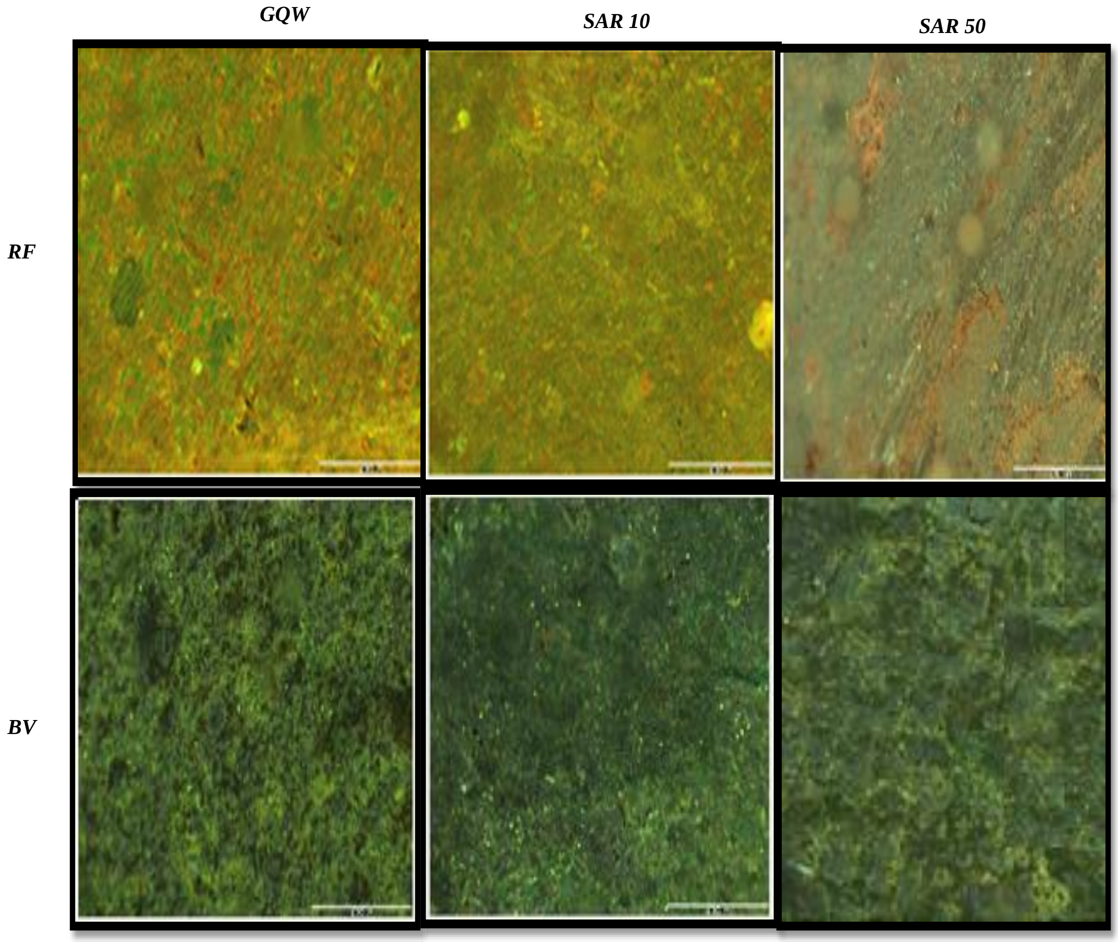


Figure 6-17 Example image of soil surface showing the effect of RW on RF and BV soil packed at 1.2 g cm-3, after treatment with GQW and saline-sodic solutions.

6.5 Conclusions

This chapter has found that the soil mineralogy, packing density, quality of water applied and volume of water applied will affect the potential for clay dispersion and migration within soil columns. This in turn affected the potential for pore blockage, clay accumulation within the columns and the soil K_s. In general, the smaller the porosity and pore size, and the higher the dispersed clay content migrating through the soil, the more rapidly pores will block. For high clay content saline-sodic soil dominated by swelling clay, the majority of pore blockage occurs in the surface 0-4 cm. However, for non-swelling saline sodic clay soil, clay migration resulted in accumulation and pore blockage at a depth of 6-8 cm. As the sodicity of the soil and water was decreased there was less accumulation of dispersed clay within the column despite dispersed clay being present in the drained leachate. As the K_s measured is a function of the most restrictive layer in the columns, this has implication for considering the validity of soil columns studies simulating soil profiles in the field, and suggests that larger soil columns may need to be used when investigating changes in soil structure associated with non-saline soils and clay migration.

Chapter 7: General discussion and conclusions

Irrigation plays a major role in Australian agriculture. However, the competition for water resources in Australia has increased with increases in population and industrial usage. The recent drought has also reduced the available water supplies. Hence, there is increasing focus on the potential to use more marginal quality water for agriculture. However, the application of poor quality (i.e. saline and/or sodic) water to fine textured soils (i.e. with significant clay content) has the potential to significantly affect soil structural stability and infiltration of water.

7.1 Dispersion and pore blockage

Applications of saline-sodic irrigation water to re-packed soil cores used in this work led to significant increases in soil ESP. The density of the re-packed cores ($\rho_b = 1$ and 1.2 g cm^{-3}) was similar to the density of freshly caltivated surface soils under field condition. The increase in ESP was significantly affected by soil density with lower density soil having significantly higher ESP after leaching with saline-sodic water compared to compacted soils. This difference could be due to the low density soils having a faster rate of ion exchange between Na$^+$ in the applied solution and the exchangeable cations upon the introduction of the saline sodic-sodic irrigation water. This may be due to faster water movement within the preferential flow paths (macropores) as the lower density soils had a significantly greater macroporosity compared to the compacted soil. Soil K_s values were also significantly affected by the ion exchange, with the K_s of lower density soils reduced more during leaching compared to compacted soils. However, the reductions in K_s during leaching were larger for RF soil compared to BV soil. This could have been due to swelling property of the dominant clays (2:1) in the BV soil that resulted in smaller pore size and lower K_s due to wetting and prior to significant ion exchange.

The application of saline-sodic water produced higher soil ESP at the soil surface compared to deeper within columns. Hence, clay swelling and dispersion is more likely to occur for soil aggregates near the soil surface. Under field conditions, the impact on soil aggregate stability of entrapped air release is also likely to be more severe at the soil surface compared to underlying soil (Oster and Jayawardane, 1998). However, the analysis of soil pores showed dispersed clay moved and blocked soil pores at different depths depending on the initial soil pores size and dispersed clay concentration.

The high proportion of soil macropores (≥ 30 µm) in the non-swelling soil (RF) soil allowed dispersed clay to move down with percolated water to accumulate at a depth of < 8 cm. In compacted soil that had lower macroporosity, the soil pore blockage generally occurred near the soil surface (< 2 cm). Conversely, the impact of clay swelling on soil pore size in the BV soil prevented dispersed clay migrating within soil pores and soil pore blockage generally occurred near the soil surface. The image analyses of soil pores showed that clay migration and accumulation commonly resulted in incomplete blockage of soil macropores to reduce total areal porosity and increase the proportion of soil micropores.

Leaching of soluble salts from saline-sodic soils by RW application (especially high ESP soils) degraded the soil aggregates so that the soil aggregates were no longer visible in the image. Reducing the soil EC below 0.5 dS m^{-1} presumably caused the osmotic potential gradient between the bulk soil solution and interior solution of the soil aggregates leading to water movement into the micropores and dispersion

(Hanson et al. 1999). Under these circumstances, soils that initially had a higher macroporosity were able to leachate out the dispersed clay. The RF soil was also less affected by low electrolyte concentration compared with the BV soil. That was presumably due to the non-swelling clay and high oxide content in the RF soil (McNeal and Colemam, 1966; Yaron and Thomas, 1968: Goldberg et al. 1988 and Ben-Hur et al. 1998).

Soil hydraulic conductivity was very responsive to changes in porosity and macroporosity. Reductions in soil macropores (≥ 30 μm) by 4% and total areal porosity of 6% resulted in significant reductions in soil K_s. There was a strong positive relationship between soil K_s and soil porosity and macropores across all soils and treatments (Figure 7-1). A significant negative correlation was obtained between soil K_s and soil micropores. The relationship demonstrated that K_s is dramatically reduced when the total porosity is reduced to < 40 cm^2 cm^{-2} and when the microporosity increases to > 40% (Figure 7-1).

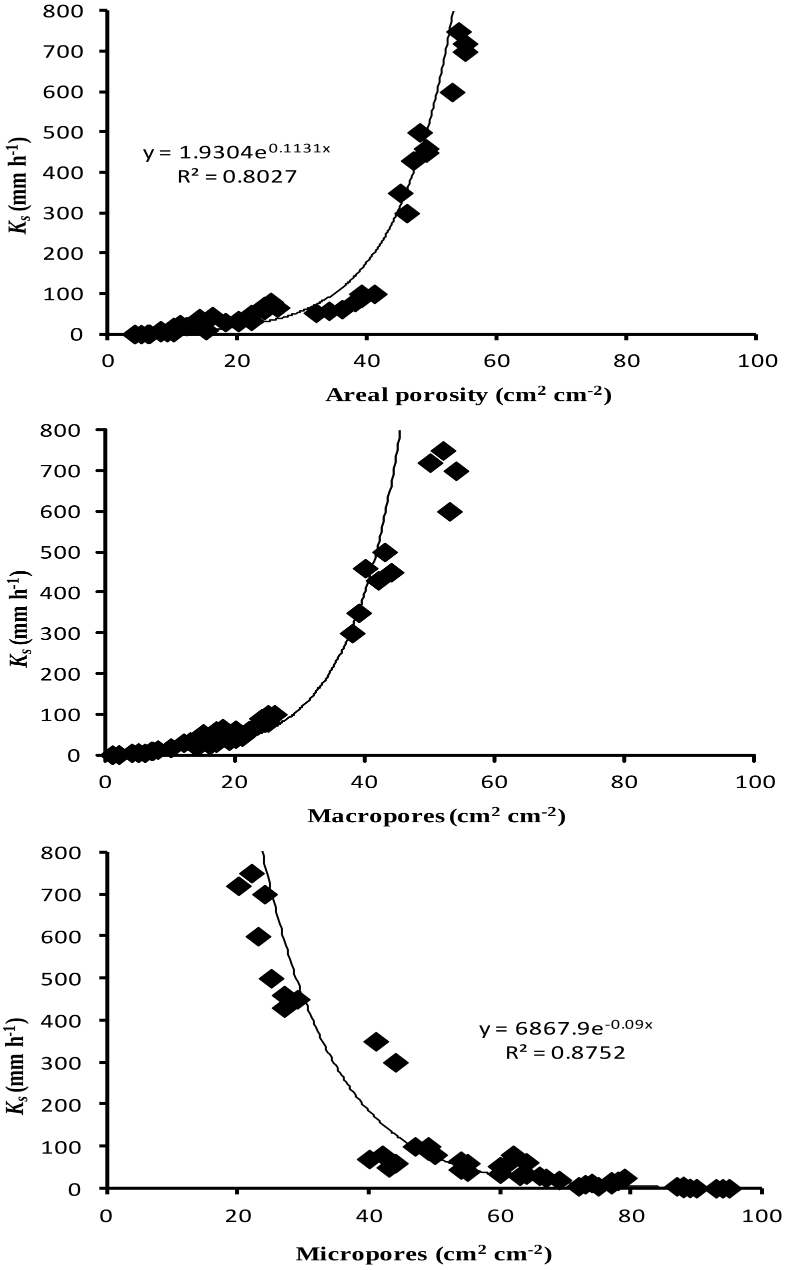


Figure 7-1 Relationship between soil porosity, pore size and Ks.

7.2 Implications for management of saline-sodic water

In this work, the equilibration of soil columns by saturation and application of 10 PV of saline-sodic water increased the soil ESP. Soil K_S was significantly reduced compared to soil irrigated with GQW due to clay swelling and dispersion. However, saline-sodic soils maintained a soil pore network that maintained soil K_S at moderate values above 15 mm h^{-1} (Schjonning, 1986; Johnson and Cramb, 1991; Snyder et al. 2000). This was most likely because the EC value of the applied water was high enough to prevent complete dispersion and structural degradation. This indicates that soils could remain permeable by maintaining the salt concentration in the soil solution above the threshold electrolyte concentration required for flocculation (Miller and Donahue, 1995; Ayers and Westcot, 1976; Barbour et al., 1998; Bauder, 2001; Bauder and Brock, 2001; Buckman and Brady, 1967; Hanson et al., 1999;

Hardy et al., 1983; Levy et al., 1999; McNeal, 1968; Oster and Schroer, 1979; Saskatchewan, 1987; Shainberg et al., 1981; Shainberg and Letey, 1984; van de Graaff and Patterson, 2001). Hanson et al (1999) suggested that, in general, soil flocculation could be maintained when the soil solution EC exceeds 1.5 dS m^{-1} or the salinity of the irrigation water exceeds 0.5 dS m^{-1}.

While increasing the salinity of the soil solution has a positive effect on enhancing or stabilizing soil aggregation, high salinity has a negative and potentially lethal effect on productivity of crops as it reduces the crop water availability (Western Fertilizer Handbook, 1995; Barbour et al., 1998; Miller and Donahue, 1995; USDA, Natural Resources Conservation Service, 2002). Thus, the EC of 2 dS m^{-1} used in this work was able to keep swelling and non-swelling soils moderately permeable even though significant reductions in soil K_s were observed compared to soil irrigated with GQW. This EC is considered slight to moderate in terms of the impact on crop water availability with EC > 2 dS m^{-1} considered to have a more severe impact consequently on crop productivity (Ayers and Westcot, 1985)

Low density soils had a lower rate of soil pore blockage compared to compacted soils even at high ESP. This was because lower density soils could percolate dispersed clay to deeper layers. Therefore, shallow cultivation to decrease soil density could be used to maintain or improve surface K_s in soil irrigated with saline-sodic water.

Considerable reductions in K_s were observed (particularly for BV soil post-irrigation with SAR 50) after exposing the saline-sodic soil columns to 4 PVs of distilled water (representing rainfall events). Soil K_s of less than 1 mm h^{-1} were observed due to the reduction in the soil solution EC below 0.5 dS m^{-1}. Mihas et al. (1998) suggested that re-irrigation with saline-sodic irrigation water increased soil K_s after rain events due to increasing the EC of soil solutions. However, further research would be required to determine if this would be the case for the soils and treatments studied in this work. Given the nature and extent of clay dispersion and pore blockage observed it seems unlikely that subsequent application of higher EC water would result in any significant increase in K_s. The most likely form of rehabilitation would involve tillage to create macropores followed by application of gypsum or a mixture of gypsum and lime to increase the soil solution EC while reducing ESP.

7.3 Implications for soil column studies for Ks as affected by water quality

Many researchers have used short soil columns for soil K_s measurements using saline-sodic irrigation water. For instance, a 6 cm column length was proposed as a standard method by Richards (1954). In addition, shorter soil columns (5 cm) have been used for measuring soil K_s (e.g. Grescimanno and Santis, 2004; Bagarello et al. 2005; Ghiberto et al. 2006; Dikinya et al. 2006; Bagarello et al. 2012). Results obtained in this research with 8 cm columns showed changes in soil K_s in response to water quality due to changes in soil pore size by clay swelling and dispersed clay movement.

The K_s of a soil column should be directly related to the K_s of the most restrictive layer within the column. This study has shown that for some water quality and soil treatments, clay dispersion and migration resulted in the accumulation of clay and pore blockage at a depth of 6-8 cm in the soil column. In these cases, if shorter column lengths (e.g. 5 cm) had been used then it seems likely that the dispersed clay would have been drained from the column. This would have resulted in a K_s for the short column larger than obtained in this trial. Conversely, for the low SAR treatment in this trial there was only limited pore blockage by dispersed clay, even at the 6-8 depth. However, if longer column had been used then it seems likely that dispersed clay movement and accumulation at greater depth may have produced a higher level of pore blockage and hence, a reduced K_s measurement. Clearly field soils normally have a depth much greater than that investigated in the columns. Hence, in understanding the behaviour of field soils and the effects of water quality on structural changes and soil water relations there is a need to ensure the column length selected for such laboratory studies adequately reflects the expected depth of clay migration and accumulation. It should also be noted that K_s decreases with increasing drainage and clay migration which suggests that the minimum volume of water that needs to be applied to the column to determine steady-state K_s will be a function of pore size and dispersed clay load.

Short column lengths (e.g. < 8 cm) seems reasonable for studying K_s changes for high clay soils dominated by swelling clays. Short columns may also be appropriate for other soils where the soil is compacted or when the effect of highly saline-sodic water is investigated. However, when low density non-swelling soils are investigated under conditions of clay dispersion then it may be necessary to use longer columns to ensure that the zone of maximum accumulation is observed.

7.4 Recommendations for further research

Recommendations for further research have been identified throughout the chapters. These recommendations include:

- Changes in soil K_s in relation to ion exchange rate when saline-sodic irrigation water was applied suggested that the steady state K_s could be measured as the ESP approached equilibrium with the applied water. However, there is a need to examine the conditions (if any) under which K_s changes further with additional leaching and the relationship with chemical equilibrium of the macropores.
- The water head applied above the soil columns was 1.5 cm during K_s measurement. However, further work could be conducted to determine whether the rate of clay dispersion, movement and pore blockage are affected by the infiltration rate and, hence the applied head.
- The soil pore size analysis of images obtained from the impregnated soil blocks at different depths provided data to help explain the mechanisms of dispersed clay movement, accumulation and pore blockage. However, further work is required to investigate the impact of pore shape and connectivity on soil pore blockage.
- Further research could be conducted to better determine the minimum soil column length by using a range of soil and water quality conditions.
- The results presented in this research are based on laboratory work. Additional research relating this data to field and crop growth conditions is essential.

7.5 General conclusions

The following general conclusions can be drawn from this research:

- Soil bulk density has a significant impact on the rate of ion exchange during the application of saline-sodic irrigation water. The exchange rate between Na^+ in the applied solution and exchangeable soil cations was significantly higher in low density soil compared to compacted soil. This was due to the high macroporosity in low density soil that provided preferential flow paths under saturated conditions.
- Reductions in soil K_s during leaching with saline-sodic solutions were significantly higher in lower density soil columns compared to compacted soils.
- The impact of ion exchange rate on changes in soil K_s was greater in RF soil compared to BV soil. This was due to the influence of swelling clay in BV soil which reduced pore size and K_s on wetting. In the RF soil the main mechanism of K_s reductions was clay dispersion and pore blockage associated with increasing ESP.
- The steady state K_s was approached before complete chemical equilibrium during leaching with saline-sodic solutions. This indicated that steady state K_s depended more on the chemical equilibrium within soil macropores and did not change significantly during the longer exchange times associated with soil micropore equilibration.
- Sodium exchangeable percentage increased significantly near the soil surface with application of saline-sodic water suggesting that the soil surface will be more susceptible to dispersion and degradation. However, the analysis of soil pores indicated that dispersed clay from the surface layer could move to a deeper layer depending on soil density and pore size.
- The RF soil packed at 1 g cm^{-3} and treated with saline-sodic irrigation water allowed dispersed clay to move down to 8 cm depths. In compacted soil, pore blockage was more evident near the soil surface. In RF low density soil the high proportion of soil macropores enable dispersed clay movement but in the compacted soil the higher proportion of soil micropores prevented the dispersed clay from moving deeper. Soil pore blockage was noted near the soil surface for BV due to swelling of clay particles that narrowed soil pores preventing dispersed clay from migrating to deeper layers.
- The large proportion of soil micropores found in zones of soil pore blockage was dominated by incomplete blockage of soil macropores. The inability to identify soil micropores within soil aggregates in degraded zones suggested smaller pores were blocked first by dispersed clay.
- Significant reductions in soil K_s were obtained when saline-sodic water was applied due to the impact of clay swelling and dispersed clay movement on soil pore blockage. However, soils maintained moderate permeability due to the relatively high EC of the applied water which maintained some level of clay flocculation.
- Severe reductions in soil K_s were obtained on saline-sodic soils when RW was applied. This was due to the impact of decreasing the soil solution EC below 0.5 dS m^{-1}. The low EC and high ESP produced clay dispersion and movement.
- Strong positive correlations were found between soil K_s, total areal porosity and macroporosity while soil K_s was negatively correlated with

microporosity. Soil K_s dramatically reduced when the total areal porosity is reduced to < 40 cm^2 cm^{-2} or when microporosity exceeds 40% of the total pore space.

- The results of column studies investigating the effect of water quality on soil K_s may be affected by column length. Short columns (< 5 cm) may be used where high clay soils dominated by swelling clays are investigated, especially when only investigating highly saline-sodic water impacts. However, longer columns lengths (> 8 cm) should be used in non-swelling clay soils dominated by macropores or where the effect of better quality water is investigated.

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