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The state of measuring, diagnosing, ameliorating and managing solute effects in irrigated systems

Freeman J. Cook, Nihal S. Jayawardane, David W. Rassam, Evan W. Christen, John W. Hornbuckle, Richard J. Stirzaker, Keith L. Bristow and Tapas K. Biswas

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Acknowledgements

In compiling this review paper we have drawn extensively from previous published reviews especially international FAO publications and that of Nadler (2005) on salinity and sodicity management, as acknowledged in the text.

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1. Introduction

Irrigation is a human devised process for changing the natural water balance at a point in the landscape with the desire to enhance the plant production. In addition to the water that is applied solutes are also applied. These additional solutes which can come from various sources are often just dissolved in the irrigation water. Solutes can often be intentionally added to the water as a means of delivering nutrients to the plant, this is often termed fertigation (Solaimalai et al., 2005). Irrigation also leads to transport of solutes in the soil and landscape systems as the additional water flux provided by irrigation will result in changed patterns of solute distribution in soils and landscapes. These solutes are often intentionally added to the soil in the form of agrichemicals (including fertilisers), or were initially in the soil and are redistributed by irrigation.

Irrigation is usually required for agriculture in regions where water is in limited supply due to the lack of precipitation during some part of the growing season. In arid climates the soils are often only inadequately drained and solutes may have accumulated in these soils from precipitation, even though solute concentrations in precipitation are low. When the additional water is added by irrigation to such soils salts are mobilised and can lead to the problem of salination of the landscape. Salination of soils globally results in the loss of 20 Mha of land per annum (Heuperman et al., 2002). Constant leaching of salt in irrigated soils is required if salination is to be prevented. This means that the salt has to be transported elsewhere, and can result in contamination of surface or groundwater.

However, when we leach the salts out of the soil we also leach other solutes, such as agrochemicals. These agrochemicals, can have detrimental effects to the receiving environments, causing changes in water quality, habitat and species composition. The time course for this process of change can mean that the consequences of irrigation are not seen until some decades or more after the initiation of irrigation and after investment in social and economic infrastructure. To improve the performance of irrigation systems there is a need to be able to diagnose how solutes will be altered by irrigation practices, measure the distribution and fate of the solutes, use the solutes for the best environmental and agricultural outcome, and ameliorate existing and residual solutes.

For irrigated agriculture to become a sustainable practice the solutes in irrigation systems must be managed. This report provides some state of the art tools to allow this management to occur.

2. Dimensionality and Scaling

Initially we will assess the dimensionality of the system we are dealing with if we are to correctly, diagnose, measure, use and ameliorate any system. Without such a process we can often not measure the correct parameter and/or repeat the same measurement without knowingly doing so. Dimensional analysis and casting the system in dimensionless variables and parameters will usually avoid these problems.

Because irrigation systems are dynamic time is one dimension that is always involved. Depending on the process involved we can often relate the actual time of the system to some other time related property and express the system in some dimensionless time variable. Examples of these time related variables in the infiltration of water into the soil are the geometric (tgeom) and gravitational time (tg) variables defined in Philip (1969):

$$t_g = \left(\frac{S}{K_i - K_o} \right)^2$$
$$t_{geom} = \left(\frac{r[\theta_i - \theta_o]}{S} \right)^2 \qquad (1)$$

where S is the sorptivity [L T-1/2], θ is the volumetric water content [L3 L-3], the subscripts i and o are respectively the values at the intake surface and initially, K is hydraulic conductivity [L T-1] and r is the mean curvature of the intake surface. Specific use of these constants in analysing disc permeameter measurements is given in Cook and Breoren (1994). Mixing, resident and residual times are also important when looking at the displacement of solutes in groundwater systems (Raats, 1981; Bear, 1972)

The three spatial dimensions are important in irrigation systems. For infiltration the process where a uniform concentration (flood) or flux (sprinkler) of water is applied at the soil surface can be treated as occurring in one spatial dimension (depth), even though the actual system is usually two dimensional with an advancing flood front and moving irrigators. The spatial dimension can then often be scaled by the total amount of water applied to give a dimensionless system of equations (Figure 1).

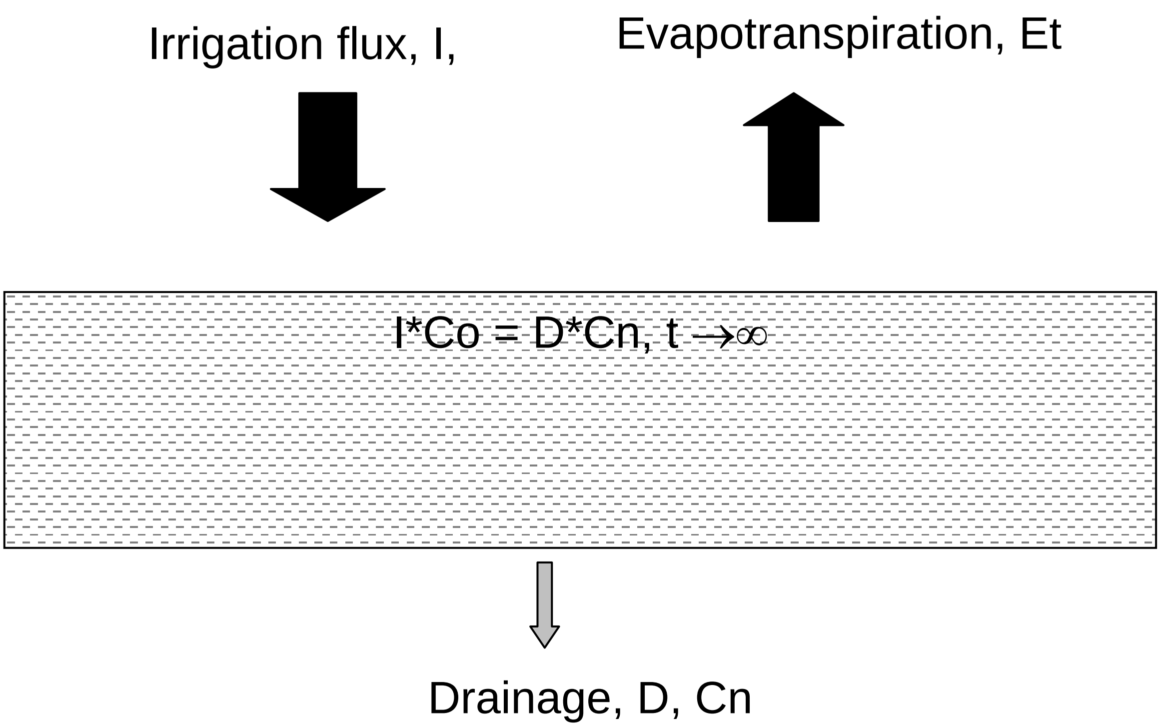

Figure 1. Simplified schematic diagram of one dimensional irrigation of soil and the accompanying salt balance. This assumes adequate drainage of the soil and no accession of water from the water table. I is infiltration flux [L T^{-1}], Et is evapotranspiration [L T^{-1}], D is the drainage flux [L T^{-1}], t is time [T], Co is the solute concentration in the irrigation water [M L^{-3}] and Cn is the concentration of solute in the drainage water [M L^{-3}].

Irrigation systems that apply water in two dimensional wetting process are common, and include furrow, trickle, sub-surface (buried trickle, sub-irrigation etc) and spray systems (particularly micro-spray) where there is no uniform coverage of the soil surface. These are often axisymetric systems where a plane can be drawn through what appears to be a three-dimensional system that turns it into a 2-D system (Figure 2). The infiltration process for solute transport from trickle irrigation has been analysed by Clothier and co-workers in a number of publications since 1984 (Clothier, 1984; Clothier and Green, 1997; Clothier et al., 1991). Similarly for transport of solutes to drains Jury (1975a,b) and Raats (1978a,b) have provided useful solutions based on advection only. More recently numerical models such as HYDUS2D (Simunek et al., 1999; Rassam et al. 2003) provide the means to simulate water and solute flow in soils for problems where the boundary conditions are mixed.

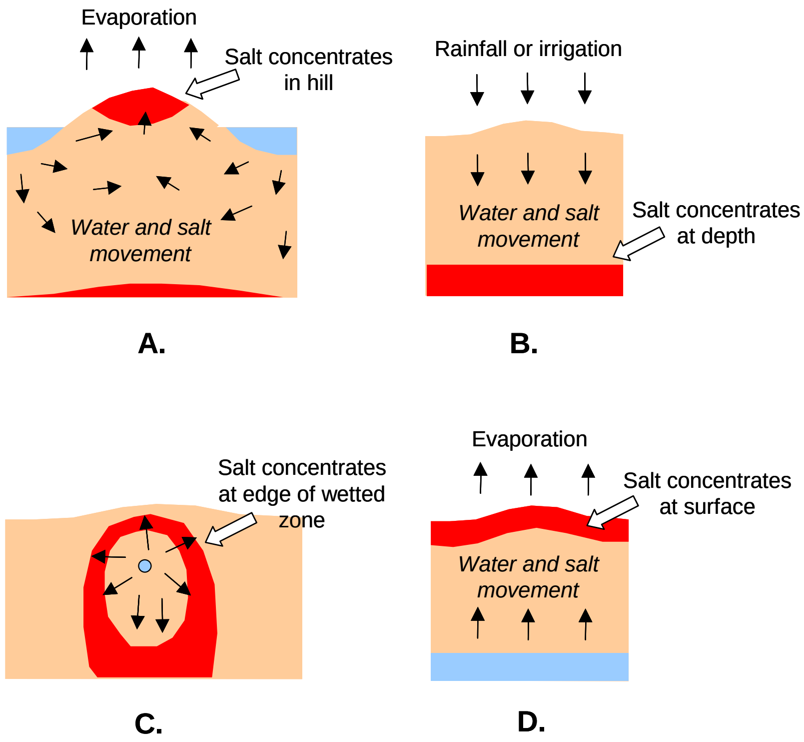

Figure 2. Typical salt distributions in A) furrow irrigated systems, B) Overhead irrigation and rainfall dominated systems, C) subsurface trickle irrigation systems, and D) situations influenced by shallow saline water tables (after Nelson 2001).

Irrigation systems that require all three spatial dimensions are rare. These include trickle and spray systems where the patterns overlap along only one axis but not the other (Figure 3).

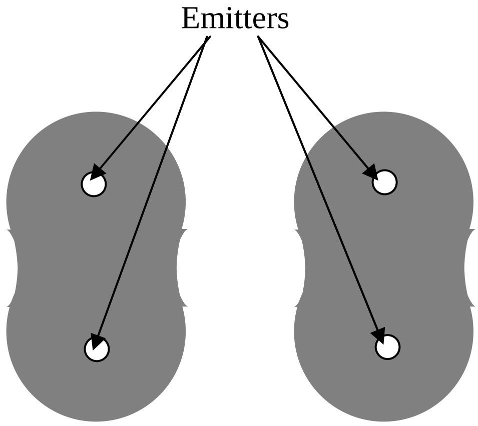

Figure 3. Schematic diagram of overlapping wetting patterns along one axis only. This is a three dimensional flow problem.

When considering solutes in the soil/vadose/groundwater system the concentration can usually be presented in scaled form which allows a reduction to:

$$C^* = \frac{C - C_o}{C_c - C_o} \tag{2}$$

where C^* is the scaled concentration, C is the measured or modelled concentration [M L^{-3}], C_o is the initial or irrigation water concentration of the system, and C_c is some 'critical' concentration. Use of such a scaled concentration will allow water from different sources to be assessed in a unified way. It is imperative that the values of C_o and C_c be presented with the results so that the value of C can be recovered if it is needed.

3. Effects of Solutes on Soil Properties and Irrigated Crop Growth

Accumulation of solutes can adversely affect the soil properties due to increasing soil sodicity leading to reductions in soil macroporosity and associated soil permeability properties. Solute accumulation can also lead to increased soil salinity which can adversely effect crop growth. These effects are discussed below.

3.1 Soil structural properties as related to salinity and sodicity

Structural stability, in soils which have a reactive clay content, is dependent on the interaction between soil sodicity and salt concentration in the soil solution. The primary processes responsible for soil structure degradation are soil clay swelling and dispersion. Clays will swell and disperse spontaneously at a given soil exchangeable sodium percentage (ESP) value when the salt concentration in the soil water is below a critical flocculation concentration, defined as the threshold concentration (Quirk and Schofield 1955). The soil ESP is closely related to the sodium absorption ratio (SAR) of the applied irrigation water. SAR is a measure of relative sodium content in irrigation waters, and is expressed as:

$$\mathrm{SAR} = \frac{[\mathrm{Na}]}{\sqrt{[\mathrm{Ca}]+[\mathrm{Mg}]}} \tag{3}$$

where [Na], [Ca] and [Mg] are concentrations in millimoles per litre (mmol L^{-1}).

Laboratory and field studies have shown that a reduction in electrolyte concentration or an increase in sodium absorption ratio (SAR) of a percolating solution results in an increase in reactive clay swelling and dispersion (McNeal *et al.* 1966), a change in pore size distribution (Jayawardane and Beattie 1979) and a decrease in saturated conductivity of soils (Quirk and Schofield 1955; McNeal and Coleman 1966). Quirk and Schofield (1955) defined the threshold electrolyte concentration, as the concentration at which a 20 % reduction in the soil hydraulic conductivity occurs, at any given exchangeable sodium percentage (ESP). Jayawardane (1977) introduced the equivalent salt solutions concept, and used this concept to predict the changes in saturated and unsaturated hydraulic conductivity of soils in the presence of different salt solutions with varying SAR and salt concentrations (Jayawardane 1979, Jayawardane 1983, Jayawardane 1992, Jayawardane and Blackwell 1991). The predicted values of hydraulic parameters could then be used in existing water flow models for assessing water and solute flow through irrigated soils.

An effect of irrigating with water having excess sodium or very low salt concentrations is the development of infiltration and percolation problems, which occurs very quickly during irrigation. The reduction of infiltration and percolation can be caused by processes such as the dispersion and migration of clay minerals into soil pores, the swelling of expandable clays and crust formation at the soil surface. The potential for occurrence of infiltration and percolation problems are normally evaluated on the basis of the salinity and sodicity content of the irrigation water. The other factor that needs to take into account is the change in solubility of calcium in the upper soil layers, during and after irrigation. The solubility of calcium carbonate in the root zone is influenced by dissolved carbon dioxide concentration, concentration of the solution and the presence of carbonates, bicarbonates and sulphates. In such calcic soils, Ayers and Westcot (1985) proposed a procedure for adjusting the calcium concentration of the irrigation water, to the expected equilibrium value following irrigation. The presence of carbonates and bicarbonates in the irrigation water could also contribute to soil degradation in the long term, because the precipitation of calcium carbonate can increase soil SAR. Calcium in the form of calcite is one of the first salts to precipitate. Upon further concentration magnesium salts will also precipitate (Tanji and Kielen, 2002).

Infiltration problems caused by the sodicity of irrigation water also depends on soil management practices adopted, such as tillage and irrigation practices used (Tanji and Kielen, 2002). When chemical bonding is sufficiently weakened by sodicity effects, spontaneous dispersion may take place. But inputs of energy can aid the dispersion process. For example, sprinkler irrigation increases the likelihood of surface crusting due to the high physical disruption as the drops hit the soil surface aggregates, which are weakened by the sodicity effects. Improved soil management practices such as incorporating organic matter increases the stability of the soil aggregates and reduces the potential for structural degradation in sodic soils.

Various researchers have developed soil stability indicators in relation to the total salinity concentration and the SAR of the irrigation water applied. The actual line that represents the division between stable and unstable soil conditions is determined by the soil type, and varies with other soil properties (McNeal and Coleman 1966, Rengasamy et al. 1984). Therefore, published generalised guidelines (Figure 4) on infiltration problems in relation to the SAR and the salinity of the applied water can provide only approximate guidance.

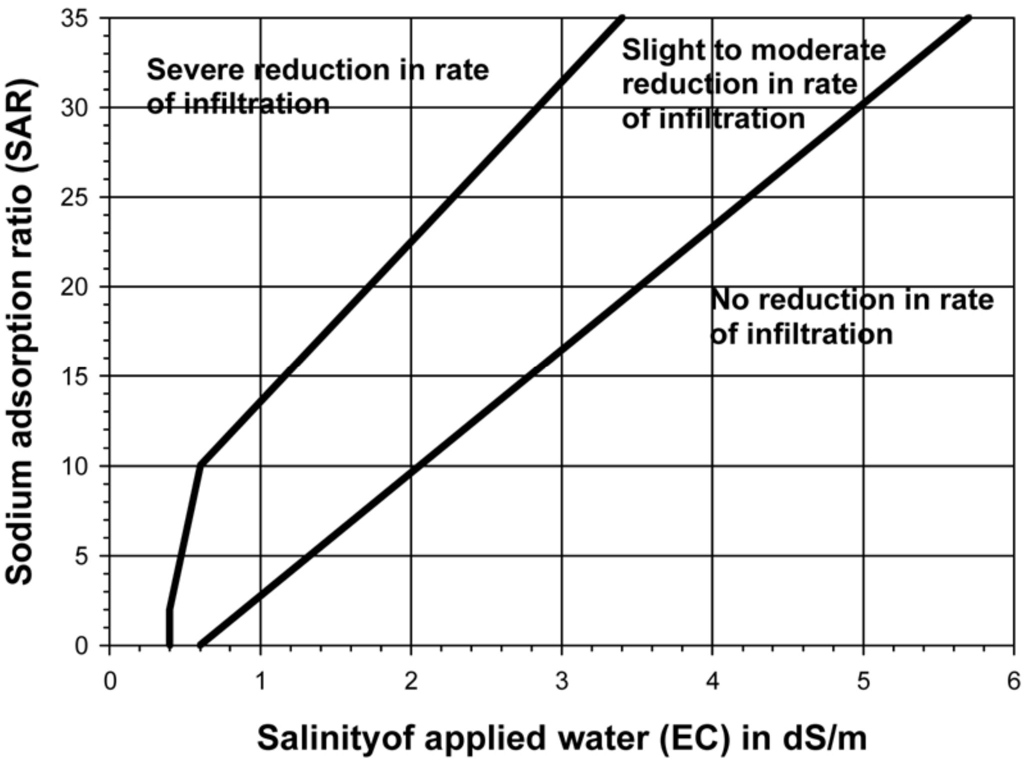

Figure 4. Relative rate of water infiltration as affected by salinity and *SAR*. Source: Ayers and Westcot (1985).

Where large-scale reuse of wastewater and drainage water is planned and sodium hazards might be expected, soil stability lines need to be established for local soils. In studies at Tatura in Victoria, the combined effect of sodium content and salt concentration on soil structural stability was quantified by Rengasamy *et al.* (1984) for a red-brown earth with illitic clays, where free lime is not present. They used the "Threshold Electrolyte Concentration (TEC)", defined as the salt concentration above which the soil will remain flocculated at a specified sodium content. They found that the TEC for spontaneous clay dispersion is given by eqn 4, where the SAR was measured in 1:5 soil:water extracts.

$$\text{TEC} = 0.016\ \text{SAR} + 0.014 \tag{4}$$

When the soil is subject to mechanical activity such as tillage and trafficking, the TEC for maintaining soil structural stability is higher than the TEC for spontaneous dispersion. Rengasamy *et al.* (1984) proposed eqn (5) to calculate the TEC for mechanical dispersion in combined surface and subsurface layers of red-brown earths.

$$TEC = 0.146\ SAR - 0.15 \quad (5)$$

He also found that eqns (4) and (5) could be used to more accurately describe the TEC for mechanical dispersion in the surface and subsurface layers of the Red-brown Earths, respectively.

$$TEC = 0.121\ SAR - 0.33 \quad (6)$$

$$TEC = 0.319\ SAR - 0.17 \quad (7)$$

Rengasamy et al. (1984) found that these equations could be used to predict the probable dispersive behaviour of the surface layers and exposed subsoil of the red-brown earths. They also used this information to provide a framework for the formulation of appropriate management strategies to maintain soil stability, based on a characterization of the soil solution composition. Jayawardane et al. (2001) used the above equations to predict the soil management needs to maintain the soil stability on a sodic clay soil used for land treatment of saline sewage effluent, under the land FILTER (Filtration and Irrigated cropping for Land Treatment and Effluent Reuse) system (Jayawardane, 1995).

The basic relationships of soil stability to the interaction between soil salinity and soil sodicity which are described above, can be considerably modified by the changes in other soil properties (McNeal and Coleman, 1966, Rengasamy et al., 1984). These other properties include the presence of iron and aluminium oxides, soil pH, presence of calcium carbonate, exchangeable Ca:Mg ratios (Emerson, 1983), organic matter, (Rengasamy et al. 1984)), and severity of drying events (Collis-George and Smiles, 1963)

In ameliorating and managing the adverse sodicity effects of irrigated soils, the appropriate techniques to be adopted need to be directed towards overcoming the specific limitations in the root zone that lower crop performance, using concepts such as the non-limiting water range (Jayawardane and Chan, 1994, Oster and Jayawardane, 1998). In terms of practical requirements of irrigated sodic soil management, there is little need to undertake action to increase infiltration and/or soil hydraulic conductivity unless water intake to match crop water needs or leaching requirements cannot be met, or if secondary problems that reduce crop yield or impede seedling emergence occur (Ayers and Westcot, 1985). Secondary problems include crusting of seed beds, excessive weed growth and surface water ponding that leads to poor aeration which can cause root rot, diseases, nutritional disorders and poor germination. Management options to address these problems include chemical, biological and physical methods (Ayers and Westcot, 1985, Tanji and Kielen, 2002). Physical methods include cultural practices to increase macroporosity and infiltration rates during irrigation and rainfall. However, to obtain long-term improvement the increased porosity created by physical methods needs to be simultaneously stabilised by chemical methods to minimise settling and re-compaction, followed by adoption of biological methods to further stabilise the soil (Jayawardane and Chan, 1994, Oster and Jayawardane, 1998). Chemical management options involve adding chemical amendments to soil or water, thereby changing the soil or water chemistry to maintain the balance between sodicity and salt concentration effects (Rengasamy et al., 1984). The aim of biological methods is to improve soil structure, or to influence the soil chemistry and stability through the addition of organic materials. Segmental soil improvement practices such as gypsum slotting can increase the longevity of the physically-induced porosity improvements in sodic soil (Jayawardane and Blackwell, 1985, Jayawardane et al., 1995) by protecting the soil from resettling and recompaction under trafficking (Blackwell et al., 1989a, Blackwell et al.,1989b). Other conservation soil tillage practices such as minimum tillage and bed farming can also minimise compaction effects on infiltration and percolation properties in sodic and clay soils (Lal, 1995, Olsson et al., 1995).

3.2 Effects of soil salinity and solutes in irrigation waters on crop growth

A primary concern in the use of saline irrigation waters is the buildup of salts in the rootzone, to an extent that it interferes with optimal crop growth and yields. These issues are discussed in detail in Tanji and Kielen, (2002), and summarised below. Applying more water than is required for crop evapotranspiration can leach the excess salts and prevent their accumulation in the root zone. Rainfall can also contribute to leaching of salt. This excess water application requirement is referred to as the leaching requirement (see section 4.1). However, the excess water flowing beyond the root zone can contribute to water table rise and potentially carry salts into the underlying aquifer, thereby contributing to environmental degradation. Hence the leaching fraction needs to be minimised. The hydrological conditions of the irrigated area should have the capacity to manage the excess leaching fraction through removal by the regional hydrological flow processes. In areas with insufficient natural drainage, excess leaching water will need to be removed through artificial drainage. This needs to be combined with appropriate saline drainage water discharge strategies.

In some soils there will not be leaching with application of excess irrigation water. Research on deep rooted crops such as lucerne on heavy duplex soils in the Shepparton Irrigation Area has shown that there may be very little or no drainage resulting from irrigation, Noble et al. (1987). In this case the salts accumulated in the profile until they are leached out by winter rainfall. Increasing irrigation amounts during the summer irrigation period would also leach salt from the profile.

Salinity in the rootzone decreases the osmotic potential in the soil solution. This causes plants to exert more energy to take up soil water to meet their evapotranspiration requirement. At certain soil-profile salt concentrations, plant roots will not be able to generate enough force to extract water from the soil profile. Water stress will then occur, resulting in reduction of crop growth and yield. The extent to which the plants are able to tolerate soil salinity differs among crop species and varieties, and is discussed in detail in Tanji and Kielen, (2002).

The variations among crops in salt tolerance characteristics are attributable to the fact that certain crops can make the necessary osmotic adjustment to enable them to extract more water from saline soils. This adjustment involves two mechanisms, namely absorption of salts from the soil solution, and the synthesis of organic solutes. Halophytes tend to absorb salts and impound them in the vacuoles. Organic solutes serve the function of osmotic adjustment in the cytoplasm. Most cropping plants are mesophytes and tend to exclude sodium and chloride ions. As a result, they are more salt sensitive than halophytes. Recent research on separating osmotic and matric potential effects shows promise in determining the osmotic impact on plants (Vetterlein and Jahn, 2004).

Sensitivity to salts changes considerably during plant growth and development. Most crops are sensitive to salinity during emergence and early development. Once established, many crops become increasingly tolerant during the later stages of growth to irrigation with higher salinity waters. There is general agreement that when plants are stressed at the early stages, there is greater reduction in vegetative growth (Maas and Grattan, 1999).

The extent to which crops suffer from salinity stress also depends on several other factors. Although yield reductions are defined as a function of the average salt concentration in the rootzone, interactions between the soil, water and climatic conditions influence the relationship.

When the crop tolerance to soil salinity and the salinity of the irrigation water are known, the leaching requirement to maintain maximum yields can be calculated. Abrol et al. (1988) proposed a model to predict the equilibrium soil salinity values at different depths in the soil profile for different combinations of irrigation water salinities and leaching fractions applied.

Selection of suitable crops and cropping practices with saline subsurface drainage waters are discussed in Tanji and Kielen, (2002). Specific Australian literature for new salt tolerant crops such as NyPa forage (*Distichlis spicata var. yensen-4a*) and Chicory *(Cichorium intybus* cv. Puna) is limited. NyPa is a perennial salt loving forage grass that is the result of selection from the native american grass. Currently NyPa Forage is being assessed across Victoria, South Australia and Western Australia. Chicory is a potentially useful forage species for dairy cows reported anecdotally to have moderate salt tolerance. These results suggest that chicory has a degree of salt tolerance that is similar to lucerne and therefore could be an alternative dairy forage species in moderately saline areas and on farms that use pumped saline groundwater for irrigation. It is also suitable for areas too acidic for lucerne. However, further assessments of its salt tolerance under field conditions are recommended (Boyd and Rogers, 2004).

Subsurface drainage waters which are sometimes reused for irrigation can also contain relatively high concentrations of trace elements, leached from the irrigated soils during flow through the soil profile. Trace elements such as boron can interfere with optimal crop growth. Other trace elements such as selenium and arsenic can enter the food chain when crops are irrigated with water containing high concentrations of these trace elements. This could be a major concern for humans and animals feeding on the irrigated crops. The site-specific nature of the practical management issues involved with boron (which is the most commonly occurring trace element) is discussed in detail in Tanji and Kielen, (2002).

4. Modelling

Models provide useful tools: for examining the processes involved in solute transport; gaining insight into how different irrigation, soil, plant and farm management systems may affect solute transport; predicting of solute transport for experimental planning; and analysing and interpreting the results of experiments. Models though are a simplification of the complex processes occurring and the results should be treated with caution. The rise of the sophistication of graphic interfaces has led to an often uncritical appreciation of the uncertainty associated with models (Cook et al., 2005b). We do not intend to recommend any particular model here as each has its applications. Below is not a review of models but discusses some models that are available to be used to consider problems associated with transport of solutes in soils.

Here we will look at two main chemical types (1) a non-reactive chemical, such as chloride and (2) a chemical that is a reactive chemical such as nitrate. Nitrate in short term processes such as infiltration will be transported similarly to chloride but over a longer time period nitrate will be taken up by the plant and other processes such as; immobilisation and mineralisation into and out of organic components of the soil will occur, and denitrification can occur. An additional chemical type occurs where the chemical species is retarded either by physical or chemical processes on its transport through the soil. This will not be considered in this review.

The main features to draw from these two chemical types are that the non-reactive type scales linearly with water transport, while the reactive solute will have a non-linear coupling with water transport. This means that the principle of super-positioning can be used for the non-reactive solute when flow processes overlap such as with trickle irrigation or groundwater systems, but this cannot be used for the reactive solute, as the concentration will have effects on the reaction rate and hence will not be simply additive.

For a rigid soil, water and solutes transport in soil can be described by the Richards equation and the convective, dispersion equation (CDE) (Hillel, 1980). These equations can be solved for simplified scenarios by analytical methods or for situations often closer to real situations using numerical methods. The spatial dimensions of the system provide a way to categorise these models.

An alternative solute transport model based on the probability density function (PDF) of travel times has also been proposed and used (Tseng and Jury, 1994; Ekard et al. 2004). This model requires some knowledge of the PDF or experimental values from which the PDF can be constructed. Such models use a probabilistic approach to the problem of preferential flow. We will not discuss this type of model any further here.

4.1 One dimensional models (1-D)

These models generally only solve water and solute flow in the vertical direction, with the assumption that the flow processes are uniform in the other spatial dimensions. For irrigation this is considered to be the case for systems where the water application is considered to be uniform such as flooded basin, border-dike and large area spray systems (rarely occurs in reality). Some of these models also consider the soil to be uniform in the horizontal spatial dimensions but not vertically where layering is allowed in some models.

The temporal dimension of these models depends on the purpose of the model. There are a number which have a fixed time step of one day. These are often tipping bucket models such as SoilWat (Probert et al., 1998; Keating et al., 2003), PERFECT (Littleboy et al., 1989; 1992), SWAGMAN (Meyer et al., 1996). Others purport to solve Richards equation but because of the fixed time step are in reality tipping bucket models with a variable transfer function between the buckets, such as WAVES (Dawes and Short, 1993), LEACHM (Hutson and Wagnet, 1995a,b). The models with truly variable time steps are numerical models like SWIM (Verburg, 1997) and HYDRUS1-D (Simunek, 1998).

There are also a number of analytical models for simplified systems where the boundary conditions are fixed or the process is considered to be some smoothed function with time. Raats (1981) gave an excellent review of analytical solutions to water and solute flow in soil with particular emphasis on residence times. His analysis provides a very good insight into the processes of water and solute transport for passive solutes but does not seem to have widely used. This in part may be due to the 'high information density' and degree of mathematics in this paper. His work would seem to provide a resource that would be helpful to many irrigators if presented in a more easily understood format.

Almost all these models use some form of a mass conservation or salt balance equation similar to equation (8) to determine the salt accumulation in the rooting zone with time.

$$C_s^i = [C_s^{i-1} W_i + Ra\, C_{Ra} + I\, C_I + U\, C_{WT} - DC_n - R_o C_{Ro}] / W_f \qquad (8)$$

where:
$\boldsymbol{C^i_s}$ is the current salt concentration in the soil element at time ***i*** (dS/m).
$\boldsymbol{C^{i-1}_s}$ is the initial soil salt concentration and $\boldsymbol{W_i}$ is the initial total water content of the soil element (mm).
$\boldsymbol{R_a}$ is rain (mm) over the time duration being considered (i to i-1) with $\boldsymbol{C_{Ra}}$ being the salt concentration in the rain water (dS/m).
I is irrigation (mm) over the time duration being considered with $\boldsymbol{C_I}$ being the salt concentration in the irrigation water (dS/m).
U is the upflow from a water table (mm) into the soil volume of interest which has a salt concentration $\boldsymbol{C_{WT}}$ (dS/m).
D is the drainage downwards to the water table (mm) which has a salt concentration $\boldsymbol{C_n}$ (dS/m).
$\boldsymbol{R_o}$ is the runoff (mm) from the soil element with $\boldsymbol{C_{Ro}}$ being the salt concentration of the runoff (dS/m).
$\boldsymbol{W_f}$ is the final total water content of the soil element (mm).

Considering the components of the salt balance it becomes obvious that rainfall amounts in total as well as in relation to irrigation timing are critical, as are the salt loads entering the soil profile either through surface water additions (rain (R_a) and irrigation (I)) or moving into the profile by capillary rise (U) from saturated soil layers. Often the concentration in the rain is considered to be much less than the other sources of salt and is taken as zero. Plant roots within the soil can be affected by salts and nutrients within the soil solution. The physiological mechanisms that cause plant responses to salt are not totally understood with osmotic effects, toxic effects and energy needs for maintenance of cellular integrity all likely to be involved.

The SODICS model uses a conservation type of equation and the measured change in chloride profiles with time to determine the average drainage (D) rate change for a particular change in land use. Estimates of the uptake rate of water can also be estimated from the negative slope of the reciprocal of the solute concentration at a particular depth if the flow regime is steady (Raats, 1981).

4.2 Two (2-D) and three dimensional (3-D) models

The groundwater flow in 2-D systems outlined by Raats (1978a,b) have been shown to be an very good approximation of the streamtube models of Jury (1975a,b) by Cook (unpublished). If these models were coupled with modern visualization methods they would have great utility in showing the effects of management changes on solute distributions.

The analytical (direct solution of the differential equations) or quasi-analytical (these contain some functions or integrals that have to be analysed using numerical methods) models are usually written in terms of non-dimensional variables which allow rapid exploration of the parameter space. These models are usually only suitable for specific boundary conditions i.e. the drip source is considered to occur at a point (this is physically impossible) but have provided good insight into axi-symmetric (Philip, 1984; Philip, 1997; Raats, 1971; Revol *et al.*, 1997a,b; Cook *et al*., 2003a,b) and 2D flow problems (Raats, 1970; Warrick and Lomen, 1981; 1983). These models have given a very good insight into the physical processes involved in irrigation and the non-dimensional variables allow the formulation of the parameter space for the numerical simulations so that redundant simulations are not created. Clothier (1984) used streamline analysis to analyse solute distribution from a drip irrigation source and in a series of papers with co-workers expand on this initial work (Clothier and Green, 1997; Clothier et al., 1991; Elrick et al., 1987). More recent Jury et al. (2003) has looked at pulses of solutes using transfer function models. This particular work has applications to the serial biological concentrator (SBC). While the assumptions regarding process (Richards equation and CDE) and soil uniformity may reduce the applicability of these models to structured and layered soils, they play an important role in simulating rigorous validation scenarios for numerical models.

Numerical models solve the differential equations by discretisation of the spatial and temporal domains (this is covered in more detail below). Commonly used methods that exist are finite difference and finite element. Finite element methods are now mostly used in 2D flow problems. More recently the method-of-lines has also been used and is a promising new method but is still in development (Matthews *et al.* 2004a,b; Lee *et al.* 2004; Schiesser, 1991). This latter method coupled with some new scaling techniques offers promise for making layered soils computationally into a homogenous soil problem.

The HYDRUS-2D package is increasingly becoming the ‘Standard’ tool for modelling variable saturated (coupled saturated and unsaturated) flow in porous media; it is being used world wide by government agencies, consultants, and universities. The software is robust and its reliability has been proven during the past decade. It is the perfect tool for modelling water flow and solute transport under precision irrigation; its time-marching scheme allows modelling the flow problem in real time (e.g., we can investigate the spatial and temporal distribution of drippers) (Cote et al. 2003; Cook et al., 2005a; Gardenas et al. 2005). Further information and uses in irrigation research for this model are canvassed in Raine et al. (2005).

Yang et al. (2002) developed a model for predicting the movement of reactive solutes through soils during saline wastewater irrigation. The model, which is based on the HYDRUS-2D code, successfully predicted the movement of water and nitrogen to a subsurface drainage system.

Irrigation water in excess of plant needs increases root zone drainage and eventually recharges the groundwater. Large irrigation developments close to a river raise the water table forming local groundwater mounds, and lead to increased discharge to the river. The solutes that are mobilised from within and below the root zone will eventually affect the water quality of the river; there are two time lags involved. First, from root zone drainage to the groundwater table; this time lag is dependent on the depth to the groundwater table and the unsaturated hydraulic properties of the soil. Second, from recharge to impact at the discharge edge (i.e., the river); this time lag varies linearly with aquifer diffusivity and non-linearly with the distance to the discharge edge (varies with the square of the distance).

Numerical models that handle variably saturated flow and mass transport (such as HYDRUS-2D) are suitable to model these processes. However, there are limitations in terms of spatial scales. When the depth to groundwater table, and/or, the distance to the river become high, the flow domain becomes huge; such problems either involve large run times or become impossible to solve. In those cases we need to split the problem into two parts; first we model 1-dimensional unsaturated flow and transport to estimate a time series for recharge and solute concentration, then second, feed this recharge time series to a MODFLOW-type model to estimate discharge to the river.

The data requirements and operational skills for physically based numerical models are usually high. As the model's complexity increases, parameterisation becomes more difficult as the number of parameters usually increases. If accuracy is to be improved more calibration data need to be collected to refine the parameters and narrow the confidence limits. Accordingly, the cost involved in running such models will be high as more parameters and input/calibration data requires more field measurements.

To avoid the complexities associated with numerical models, a 'Rapid Assessment Approach' may be adopted. Such a technique requires the analyst to complete a thorough review of applicable measured data and models in order to determine the importance of various variables; this approach has been implemented in the SIMRAT model (Fuller et al., 2004; Rassam et al., 2004), which assesses discharge response of a single unconfined aquifer to changes in recharge occurring at some distance from the river.

SIMRAT models the wetting of an unsaturated soil profile due to increased deep zone drainage; it accounts for the spatial variability of soil's hydraulic parameters by using a lognormal distribution, which results in a smooth time series for excess recharge due to the increased deep zone drainage. The cumulative flux to a river resulting from the excess recharge applied to a semi-infinite system similar to that shown in Figure 5 is given by (Knight et al., 2002):

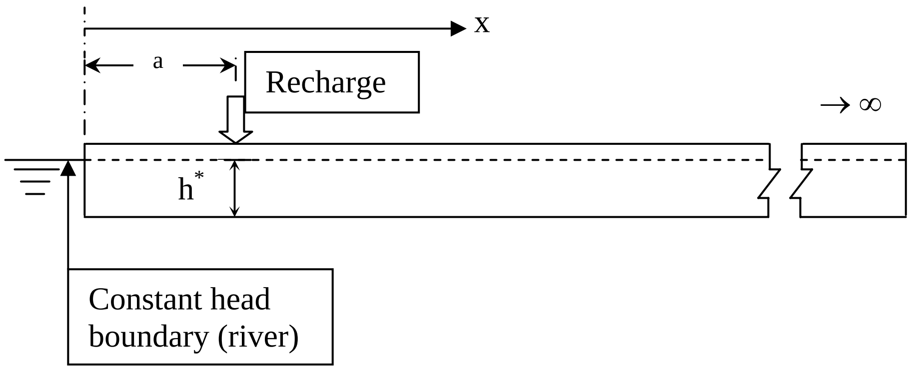

Figure 5. Recharge to a semi-infinite aquifer

$$F(a,t) = -erfc\left[\frac{a}{2\sqrt{(Dt)}}\right] \tag{9}$$

where (a) represents the distance separating the recharge source from the river (the constant head boundary), (t) is a time variable, and (D) is the diffusivity ($D=Kh^*/\phi$, where K is the saturated hydraulic conductivity, h^* is the average height of the water table, and ϕ is the specific yield of the layer where the water table exists.

Rassam et al. (2004) presented eqn (9) in dimensionless form where dimensionless time τ is, $\tau = t/(a^2/D)$. Figure (6) shows that the flux response is highly non-linear at early stages up to $\tau \approx 20$ after which it increases marginally with time. This is because the region is unbounded (semi-infinite) in the positive x-direction; initially half of the flux goes in the positive direction, and returns very slowly towards the stream (Knight et al., 2002). A key assumption in the model is linearity of the equation, which allows multiple impacts due to different actions to be superimposed. Rassam et al. (2004) provided a detailed study of the applicability of eqn (9).

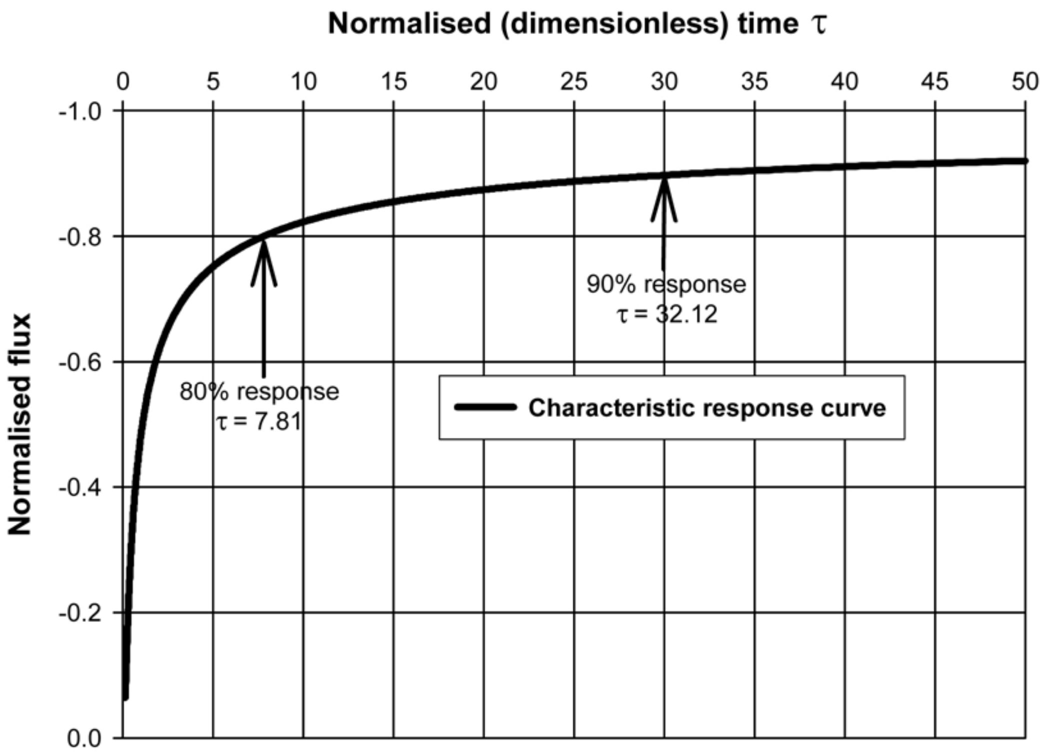

Figure 6. Non-dimensional characteristic flux response

Dillion (1988) provided an analytical solution for the flow in an unconfined aquifer with retardation due to linear adsorption and exponential decay of the solute during its passage. This allows reactive solutes to be considered but requires that the transverse dispersion of the solute is treated as a non-reactive solute, or an approximate average transverse dispersion is assumed. These models for groundwater flow all assume that the discharge of solute below the root zone is known.

5. Amelioration of Saline and Sodic Soils

5.1 Soil salinity amelioration

The amelioration and management of saline soils have been extensively studied and reviewed in previous literature (Abrol et al., 1988; Tanji and Kielen, 2002). Here we will present some recent Australian experiences with amelioration of solutes.

Plant roots also play a major role in soil-water and solute dynamics by modifying the water and solute uptake patterns in the rooting zone. Clothier and Green (1997) considered them to be the major factor affecting the transport of water and solutes in soils. Mmolawa and Or (2000) observed that roots complicate the water and solute distribution, as often the root distribution is not known and the root uptake patterns are highly dynamic. This uptake pattern can be further complicated by the osmotic effects due to salinity. Stirzaker and Passioura (1996) showed that the build up of salt around the root even in solutions that would not be considered saline can effect the transpiration of plants.

The potential for managing root zone salinity and the application of leaching fractions is increasingly important as precision irrigation is implemented. Stevens *et al.* (2004) reported soil salinity data measured on 20 citrus and grape vine sites located in the Riverland and Sunraysia regions. The electrical conductivity of the applied water was low (<0.4 dS/m) and irrigation management typically resulted in 15-20% of the applied water contributing to deep drainage which should have been sufficient to main salt levels in the root zone below plant tolerance levels. However, they found that the upper range of average EC_e in Sunraysia sites was above the threshold for salinity damage to vines and in the Riverland above the threshold for both vines and citrus. The calculated mean leaching efficiency of 0.63 at these sites was significantly less than unity ($P < 0.01$) and had a large coefficient of variation (77%).

The use of particular species of plants as indicator (canary in the mine) type of plants may be a potential diagnostic tool. Roses have been used in vineyards as such indicator plants with regard to downy mildew. They suffered the effects of this fungi well before grapevines, alerting growers to spray for mildew before it effects the grapevines. It may be possible to use plants that are susceptible to salinity in the same way. However, the mixing processes are much slower in the soil than the atmosphere, so this may not be practical.

5.2 Leaching Fractions

The amount of solute (salt) in the root zone of a crop needs to be lower than some threshold and should remain constant if the crop is not to be detrimentally effected and yield reduced. This has led to the concept of a leaching fraction (L), where a certain amount of water in excess of evaporative and transpirational needs is required to maintain the steady solute concentration

$$L = \frac{D}{Ra + I} \approx \frac{C_o}{C_n} \tag{10}$$

where $C_o = \left(RaC_{Ra} + IC_I\right)/\left(Ra + I\right)$ and other terms are defined earlier. This assumes that solute is displaced from the root zone as piston flow which is not always the true. Stevens (2002) and Stevens et al. (2004) have questioned the assumption of 100% leaching efficiency.

Where partial root zone wetting or non-spatially uniform wetting occurs (micro-spray, trickle) during irrigation the drainage rate will vary spatially and the concept of a leaching fraction as in eqn (4) is no longer valid. Richards (pers. comm.) has shown (Figure 7) that for trickle irrigation and partial root zone wetting solutes can build up around the perimeter of the wetting zone. These non-uniform solute distributions due to fertigation had been shown earlier by Kafkafi and Bar-Yosef (1980). Raine et al. (2005) have suggested that this behaviour is a fertile area for further research.

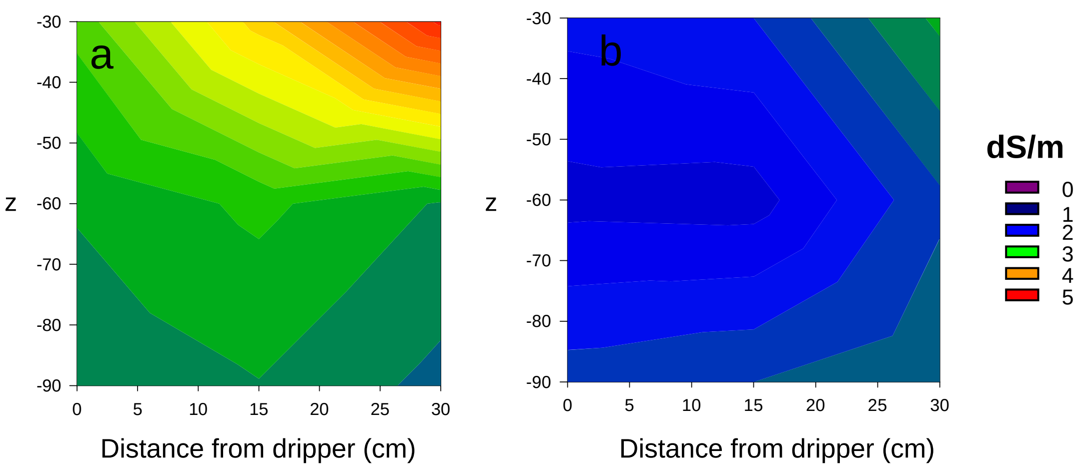

Figure 7. Electrical conductivity with depth (z cm) and distance from the dripper with drippers centred on the vine row for a) conventional drip and b) partial root zone drying (courtesy of A. Richards).

The problem of salt accumulation under a line of trees was discussed by Stirzaker et al. (1999). They calculated the maximum time between flushing events, where the salt was redistributed away from the root zone. These flushing events can be an occasion flood irrigation event or a large amount of rain. Such ideas should be explored further for trickle and other irrigation systems where salt is likely to accumulate in a non-uniform manner.

5.3 In-situ measurement of salt leaching efficiency

In the past decade many growers in the Lower Murray irrigation areas have achieved water use efficiency (WUE) of at least 85% (Stevens 2002). Water use efficiency (WUE) has been defined as the volume of water used “consumptively” by the crop i.e., evapotranspiration divided by the total volume applied to the field. Even though there is still a leaching fraction, it is unclear how efficiently salt is being displaced from the soil profile particularly in semi-arid region with low rainfall. Several growers from South Australia have reported high levels of sodium and chloride in leaf, grape and reduction in wine quality. There is also some evidence of tree mortality.

Leaching efficiency is the efficiency at which drainage water mixes with the soil solution (Bouwer 1969) and is often assumed as 100% when every millimetre (mm) of water passing below the root zone carries completely mixed soil water. A new method is proposed by (Biswas 2006) for in-situ measurement of salt leaching while watching the root zone salinity trends. The leaching efficiency can be estimated by comparing the chloride concentration in the Wetting Front Detector (WFD) with that of the soil water extractor (SWE) when installed at the same depth. The WFD is a buried funnel-shaped device used to indicate wetting front and passively collect soil water sample at <2 kPa suction (Stirzaker 2003). In contrast, SWE is a porous ceramic device that samples soil water under a suction of 60-70 kPa created by a 60 mL plastic syringe. It is assumed that the WFD is sampling both matrix and preferential flow and that the SWE is sampling the matrix (residual) soil water. This method is only valid when wetting fronts are regularly passing at the depth of measurement, such that salinity conditions in the soil above the devices are fairly uniform. Leaching efficiency, expressed as percentage, can be written as:

$$LE = [1 - \frac{[Cl_{SWE} - Cl_{WFD}]}{[Cl_{SWE}]}] \times 100 \qquad (11)$$

where Cl_{SWE} = Chloride concentration (mg/L) in soil water extracted by SWE
Cl_{WFD} = Chloride concentration (mg/L) in soil water captured by WFD.

Soil water EC (*ECsw*) was used as surrogate measure of Cl_{SWE} using a conversion relationship from 55 soil water samples collected from the field. The relationship was:

$$Cl_{SWE}\ (\text{mg/L}) = 348 * ECsw\ (\text{dS/m}) - 138.4;\ r^2 = 0.99\ (n=55; p=0.05) \qquad (12)$$

Equation (11) was used to calculate a complete set of LE calculated for 0.3 and 0.6m soil depths under both drip and sprinkler which is listed in Table 1. The LE values under drip varied between 48 and 85% giving an average of 65%. At the same time, the sprinkler LE estimated for the topsoil layer varied between 70 and 107% giving an average of 90%. Given that the initial hypothesis of wetting fronts passing regularly both the SWE and WFD, there were times after fertiliser application when WFD recorded higher EC readings than SWE. As a consequence LE values exceeded 100%. The result shows that LE measured during the same period for drip at 0.6m were higher than the top 0.3m layer. The average LE value for the subsurface layer (0.6m) was 79% with a range from 67% to 104%.

Table 1. A comparison of leaching efficiencies for two different systems

Irrigation Type	Irri	Rain (mm)	ETo	Soil Type	Depth (m)	Mean LE (%)	Range
Drip	593	334	1367	Light Sandy Loam	0.3	65	48-85
				Sandy Clay Loam	0.6	79	67-104
Sprinkler	598	334	1367	Fine Sandy Loam	0.3	93	70-107
				Sandy Clay Loam	0.6	NA	-

A 65% LE for the drip implies that at least one third of drainage is un-mixed irrigation water that bypassed without removing salt. This is particularly of concern when the leaching fraction is only <15% of total applied water in semi-arid irrigation districts.
Figure 8 gives seasonal changes of LE in a drip irrigated vineyard where highest LE was measured in winter when plants are inactive, soil is moist, and ET is minimum. This implies that LE of a permanent horticulture planting may be governed by the irrigation type, evapo-transpiration, rainfall and its intensity and distribution, cover crop and associated crop management practices.

Winter (June & July) seems to be the best time when a supplementary leaching irrigation is likely to maximise salt displacement from the root zone with minimum drainage volume. In a laboratory study using intact soil cores from the same site, (Kies 2006) found that using same volume of irrigation intermittent application was more efficient to leach salt than continuous (Table 2). The measured leaching of 0.3g/L extra salt by intermittent application is equivalent to approximately 0.5 ton per ha, equating about 10% increase in salt leaching.

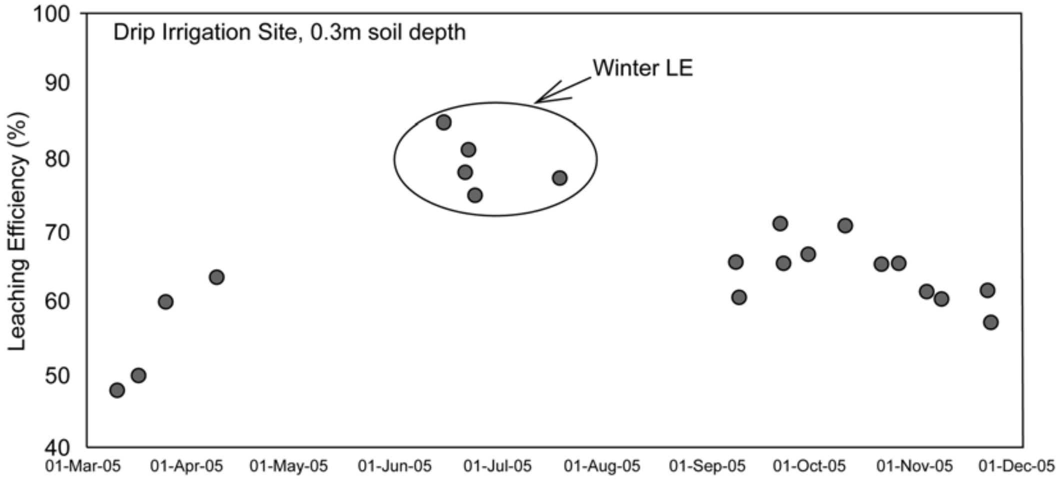

Figure 8. LE of a drip irrigated vineyard's topsoil

Table 2. Comparison of salt leaching by continuous and intermittent leaching irrigation

Core #	Total volume drained (L)	Continuous EC (dS/m)	TDS (g/L)	Intermittent EC (dS/m)	TDS (g/L)
1	3.5	2.125	4.8	2.289	5.2
2	3.5	2.097	4.8	2.237	5.1
3	3.5	2.163	4.9	2.296	5.2

5.4 Soil sodicity amelioration

The identification and agricultural management of sodic soils have been extensively studied in many countries and reviewed in previous literature (Abrol et al., 1988; Sumner and Naidu 1998, Tanji and Kielen, 2002). In Australia sodic soils occur extensively in the irrigated areas. The poor crop productivity of sodic soils is often associated with their low infiltration and restricted internal drainage (Rhoades and Loveday, 1990). This is caused by low macroporosity and macropore instability, due to presence of sodium on the clay surfaces. To achieve long-term amelioration, tillage techniques to increase macroporosity have to be combined with chemical and biological techniques to improve macropore stability, such as addition of chemical ameliorants and incorporation of organic matter (Jayawardane and Chan, 1994 and Oster and Jayawardane (1998). Maintenance of macroporosity also requires protection of the tilled soil from recompaction during flood irrigation, raindrop impact and trafficing.

In the surface layers of soil, adverse effects of sodicity can be corrected by incorporating gypsum or other sources of calcium of magnesium, and by using conservation farming practices to add organic matter and to protect the surface from mechanical disturbance and raindrop impact (Lal 1995). Subsoil sodicity can be corrected by combining deep ripping with chemical ameliorant additions, but the beneficial effects are often quickly lost under flood irrigation and trafficing. Longer term increase in crop production can be achieved by improving surface and subsurface drainage (Spoor, 1995), bed farming (Olsson et al., 1995) and gypsum slotting (Jayawardane and Blackwell 1985, Jayawardane and Blackwell 1986, Jayawardane et al. 1995). Advantages and disadvantages of these techniques, their application in dryland and irrigated cropping areas and the needs for future research was reviewed by Jayawardane and Chan (1994). In soils such as vertisols with high shrink-swell potential, crops such as safflower could be used for biological soil loosening, through deep profile drying.

The effectiveness of soil ameliorative techniques can be evaluated by assessing the soil factors limiting crop growth during a growing season in a non-ameliorated soil, and the subsequent changes in these soil factors due to the ameliorative practices. The on-field application of such tools to monitor these changes throughout the cropping season in Australian soils, based on the "non-limiting water range" concept is described in Jayawardane and Chan (1994) and Oster and Jayawardane (1998).

6. Measuring

6.1 Electromagmetic methods

Recently Nadler (2005) has given a comprehensive review of electromagnetic methods for measuring the electrical conductivity of soils, extracts and solutions. The information below closely follows that of Nadler (2005).

Except for the TDR, three kinds of portable resistivity sensors are available: (i) under-surface-installed four-electrode sensors (Rhoades and van Schilfgaarde, 1976), (ii) surface-array resistance sensors (Wenner array), and (iii) EM induction sensors (Geonics EM38) are available for measuring the soil electrical conductivity (σ_a). The four-electrode sensor is directly inserted by preaugering to the depth of interest. An accurate, direct contact σ_a value is obtained, representing a soil volume of ~0.1 L. The other two sensors are depth-weighted σ_a, and the weighting functions vary with the configuration of the electrodes, or electromagnetic coils, frequency of electrical current used in the measurement, distribution of σ_a within the various depths of the soil profile, and other factors. They provide empirical, broadly correlated, and widely scattered data. These sensors, plus some site-specific calibrations, are usually adopted as survey tools for salinity assessment, precision agriculture (Corwin and Lesch, 2003), and mapping (Lesch et al., 1992).

Linear correlation coefficients ranging from 0.61 to 0.98 were found between soil paste and Wenner arrays (Read and Cameron, 1979) with a standard deviation of up to 4 for a σ_a =7 dS m^{-1}. The EM38 instrument (Rhoades and Corwin, 1980) has been used to survey large areas to indicate the extent of salinity by measuring σ_a of the soil profile. The device sensitivity varies with depth, orientation, height from the soil surface, water content (θ), degree of homogeneity, and the interaction among all of the abovementioned parameters such that the error in average σ_a value compared with that of the resistivity four electrode probe is slightly higher (~15% according to Rhoades and Corwin, 1980), or significantly higher according to Slavich and Petterson (1990). For a depth of 0–0.6 m in a high clay soil with high salinity the calibration becomes non-linear. Sudduth et al. (2001), applying the EM38 for precision agricultural practices, found difficulties in separating the dependency of measured σ_a on electrical conductivity of the solution (σ_w), water content (θ), topsoil depth, clay content, clay mineralogy, soil pore size distribution, temperature, season, and, to a smaller extent, the variation in sensor operating speed and height and drift over time. The reason is inherent to the system. On one hand, assuming all other conditions are constant, a high clay content soil is closely associated with higher levels of, θ, σ_s (solid phase surface electrical conductivity), and salt accumulation (due to slower leaching rates).However, on the other hand, the contribution of these three parameters to σ_a cannot be separated. Moreover, their effect on σ_a will be contradicting, thus resulting in less accurate σ_s (see also Sudduth et al., 2003, and Heiniger et al., 2003). Only site-specific calibrations, often with each measurement set, enabled use of within-field σ_a data for evaluating depth of topsoil, profile-θ (integrated over the volume of soil sampled), clay content, and soil drainage class, with accuracies up to ±70%.

Relations between the aqueous extracts and resistivity obtained σ_a, converted to extract electrical conductivity (σ_{ext}) is probably the most common salinity appraisal technique, yet caution should be practiced when comparing results of these two methods. θ_{soil} changes may modify the σ_w of the soil solution by varying the amount and composition of the dissolved ions. Unless the soil is very sandy and the majority of the ions originate from soluble salts, only a limited correlation between σ_{ext} and σ_w is expected. Nadler (1997) reports the differences between dilution-corrected aqueous soil extracts (σ_w, σ_{ext}) and σ_w relations for three soil types. For sand, the relations were almost linear and well correlated (R = 0.92). σ_w, σ_{ext} data points were 0.5 – 2.0 dS m^{-1} above the 1:1 line as a result of disturbing the θ-dependent chemical equilibrium by the extraction process and carrying it over, by adjusting calculations, to lower levels. For loam soil, the relations were still linear but there was scatter (R = 0.76) and σ_w, σ_{ext} data were evenly spaced above and below the 1:1 line. For clay soil, totally different relations were found, composed of two seemingly non-related parts. In one part, for a θ increase from 0.1 to 0.4, σ_w = 1.3 ± 0.3 dS m^{-1} and the $\sigma_{w,\ ext}$ ranged from 4 to 20 dS m^{-1}. In the second part, the σ_w increase was from 2 to 6 and $\sigma_{w,\ ext}$ values fluctuated between 2.5 and 4.5 dS m^{-1}. With an increase in θ, the scatter in σ_w decreased by, a small amount in sand or larger amount in loam soil. For clay soil the decrease in scatter in σ_w only started at θ > 0.6. The expectation for a constant product $\sigma_{w,\ ext}.\theta$ at high y values contains an assumption that the solute chemical composition and the dilution effect on ionic activity coefficients (the ionic property used for estimating concentration from s measurements) are constant. However, the dilution by addition of water in the high θ levels of the sand and the clay caused a relative increase in concentration for divalent (over monovalent) ions in solutions, so that the same total salts (equivalents) content was represented by a somewhat lower $\sigma_{w,\ ext}.\theta$ value. This indicated that a simple correction for the dilution (extracting) effect is not enough to reconstruct the real salinity, particularly for higher clay contents and in the presence of slightly soluble minerals. In a field experiment where three salinity levels of irrigation waters (0.8, 1.3, and 1.7 dS m^{-1}) were used, measured salinity levels depended on the monitoring technique. Aqueous extracts ($\sigma_{w,\ ext}$) vs. the $\sigma_{four\text{-}electrode}$ consistently differed by 50% for a sandy soil (Nadler and Erner, 1998).The advantage of the resistivity results was supported by (i) a closer agreement to salt input mass balance, (ii) having more systematic salinity seasonal changes, and (iii) a more reasonable level of calculated leaching fractions.

Corwin and Plant (2005) provide an overview of various aspects of measuring apparent electrical conductivity. The review of Friedman discussed in detail the conversion of σ_a to σ_w and the various models that are available to do this. To make this conversion requires that the water content is also known. A few articles (White et al., 1994, and references therein, Sun et al., 1999) claim that θ measured with a TDR is σ_w

dependent, which complicates the conversion of σ_a into σ_w, if TDR is to be used to obtain σ_a. Nadler (2005) suggests that such conclusions may be reached from preliminary experiments, using software based on TDR trace evaluation, by not adjusting the observed endpoint of layered media, and by incorrect data interpretation that will rarely be encountered under routine agricultural practices. Harlow et al. (2003) suggest that TDT (time domain transmission) methods may not suffer this effect but are not good at measuring σ_a. Thus a combination of TDR and TDT methods may be required if σ_w is to be measured *in situ*.

At a large scale airbourne electromagnetic surveys have been used to gain estimates of salt concentrations and distributions in three dimensions (Cresswell et al., 2004). They caution that considerable effort is required to calibrate such surveys if misleading results are not to be produced. The results produced are only semi-quantitative estimates but they are still useful and provide a means of determining where further studies in a catchment are required.

6.2 Solute sampling devices

Sampling of the soil solution with suction cups and lysimeters is not a new technology but recently the technology has been modified to also sample water that is draining down the soil profile as bypass flow (Stirzaker and Hutchinson (2005).

The potential for using solute monitoring to evaluate the irrigation strategy is illustrated in Figure 8 from data collected by SARDI (Biswas et al, unpublished). Irrigation (drip) and rainfall are shown by the red and blue bars respectively, and the EC of soil water at 30 and 60 cm plotted as blue and red diamonds respectively on the right hand axis. Up to mid April the crop factor averaged around 0.6 and the EC in solution at 30 cm depth started to rise. After this time the crop factor approached 1 and the EC at 30 cm started to fall. Solute was collected for the first time at 60 cm, and the values were very high, suggesting that a salt front was being pushed through the rootzone.

In this case the data comes from wetting front detectors (passive samplers) and paints a picture of irrigation management even in the absence of water content measurements. The data from other sites was less clear. Sometimes passive samplers collect a sample in the top soil while the active samplers (suction cups) do not, and the picture is frequently reversed at depth. Active and passive lysimetry sometime produce similar EC values and at other times they diverge, suggesting that the salt is moved in pulses. Our current understanding is inadequate to explain these results. A similar case could be made for bulk EC monitoring by TDR and capacitance. The devices show great promise, but their output has not been adequately evaluated and exploited.

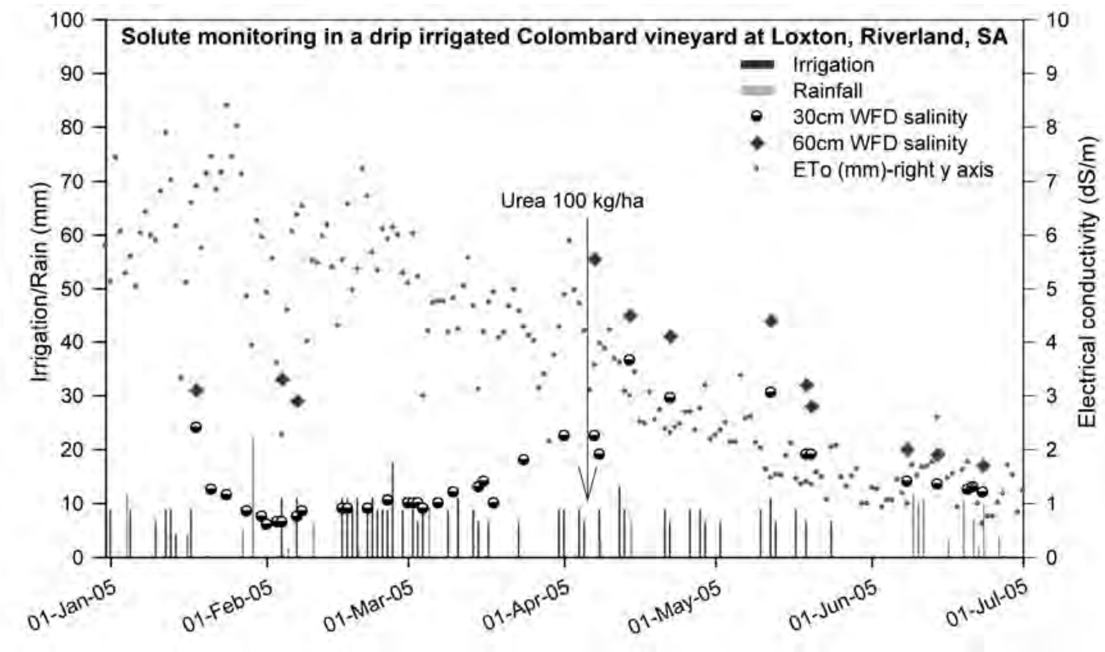

Figure 9. Irrigation, rainfall, evapotranspiration and EC measured at two depths in a drip irrigated vineyard during an irrigation season (modified after Biswas *et al.* 2006).

New application methods for fertiliser also open the door for vast improvements in nitrogen use efficiency. Stirzaker (1999) estimated the N-use efficiency for a selection of horticultural crops in Australia by comparing the application rates recommended by State agencies against the N-removal in crops. Even if farmers limited themselves to recommended rates, and obtained high yields, the efficiency of N-use would be 30 to 50%. There exists an opportunity for improving the efficiency of nitrogen use, and probably other nutrients as well.

The concepts of salt tolerance thresholds and leaching fractions are currently based around on empirical studies where one dimensional wetting, water flow and infrequent irrigation (Mass and Hoffman 1977, Ayers and Westcott 1989) was used. These approaches may not be suited to drip and micro irrigation. The type of hydrological pathway in the root zone (matrix flow or preferential flow), has a significant effect on the leaching efficiency, pesticide movement, transformations of nitrogen and nutrient losses in soils, and this introduces uncertainty in interpretation of monitoring results. Moreover there is uncertainty as to what the different tools used for monitoring solutes actually measure and the thresholds for taking management action.

Since the accumulation and leaching of different salts is inextricably linked to irrigation practice, there is a vast synergy to be gained by managing water and solutes together. It is clear that better water management will improve solute management. However, the overarching hypothesis of this program is that the monitoring of solutes will provide completely new insights into how irrigation is being carried out. Solute signatures, if properly decoded, can reveal the limits to the responsible practice of irrigation.

Active lysimetry (suction cups) has been used for decades, and although subject to considerable site to site variability (particularly for nitrate), has undergone intensive scientific scrutiny (see for example reviews by Litaor (1988) and Paramasivam et al. (1997).

Two new methods for solution monitoring have appeared in the last 5 years. First is the inference of bulk conductivity from dielectric measurements. TDR has long been used for bulk EC monitoring with some success at low solution concentrations. Vogeler et al. (2001) found that TDR was good at determining the shape of the solute front but the concentrations obtained were only a relative measure of solute concentration and direct calibration was required to get actual concentrations. More recently capacitance devices that are purported to measure EC have become commercially available. These capacitance devices give results which may allow relative concentrations to be obtained but, there remains uncertainty about the influence of soil type on the measurements. These devices do have enormous potential to assist in the management of solutes in agriculture if used wisely. Second the availability of passive lysimetry (flow distortion wetting front detector) offers a simple method of routine soil water sampling.

At this stage there is a significant problem with interpretation of data. One problem is setting a threshold from a point measurement when there is a three dimensional distribution of solute in a profile. Another is how to relate measurement from a passive lysimeter to ECe, or how to link bulk EC readings from dielectric measurements to what the plant is actually experiencing. The tools need to be evaluated side by side to help answer these questions.

6.3 Simple tool for in-situ monitoring of root zone salinity

Recognising the need for an inexpensive and simple tool for real time monitoring of soil water salinity (Biswas 2006) developed a modified porous ceramic cup device called soil water extractor (SWE), that samples soil water under a suction of 60-70 kPa created by a 60 mL plastic syringe. Although there are several devices (eg. resistance, capacitance, passive and suction lysimeters) available, these are often expensive and require specialised skills. This inexpensive device, when permanently installed, enables growers to track the salinity within the rooting depth throughout the year by sampling at any time.

Using SWE at 0.3, 0.6 and 0.9 m soil depths installed 0.15 m away from the dripper (dripper spacing 0.6m) Figure 10 presents one year root zone salinity changes within a drip irrigated vineyard growing high yielding (~30 t/ha) Colombard variety grapes in a sandy loam soil since 1985. The EC readings from SWE were found to be twice the saturated soil paste EC (ECe) having regression equation:

$$\textit{ECsw}\ \text{(dS/m)} = 1.9 \times \text{ECe (dS/m)} + 0.6,\ r^2\ 0.94\ (n=9;\ p=0.05) \qquad (13)$$

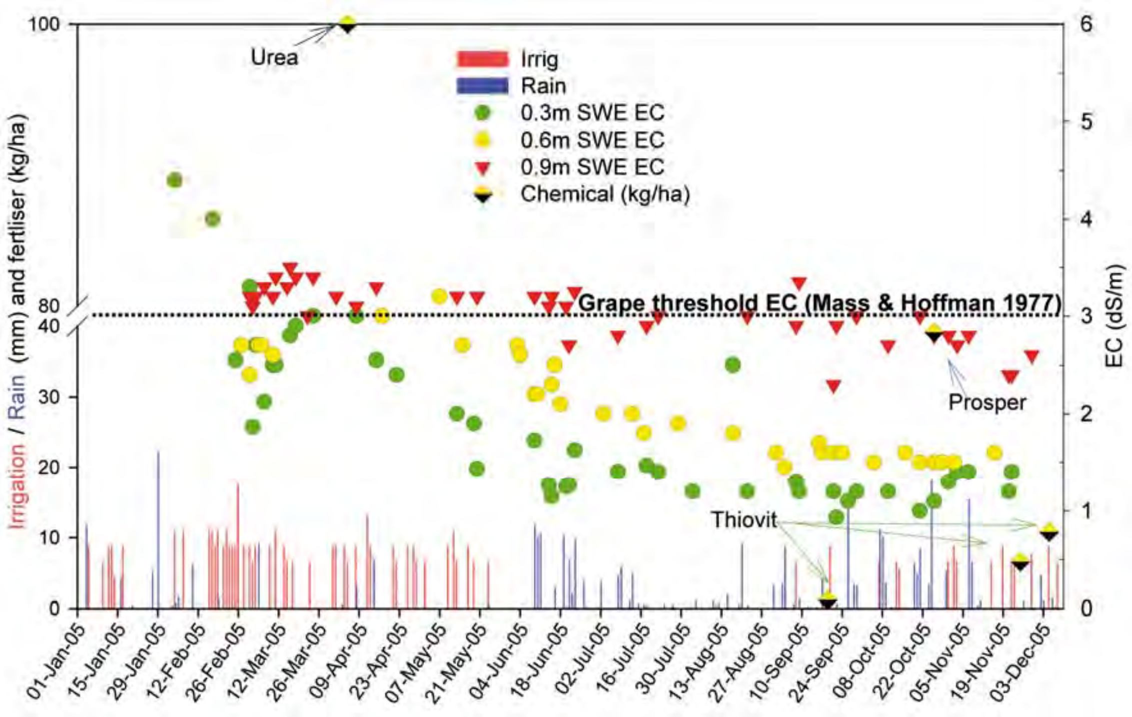

Figure 10. Salinities measured by SWE in a drip irrigated vineyard. Irrigation/rainfall, fertilisers and chemicals (left y axis); SWE salinities (right y axis).

Coupled with high water demand by plants summer irrigation increased average salinity up to 3 dS/m which fell below 2dS/m at the end of winter. The above average annual rainfall of 334 mm during 2005 in the Riverland was sufficient to leach the irrigation-induced salinity from the root zone. On a similar soil when irrigated with under canopy mini sprinkler system, a chardonnay vineyard recorded root zone salinity between 1-2 dS/m during summer irrigation period which the fell below 1 dS/m with winter rainfall. Two years of SWE data on seasonal changes of mean soil water salinities from Sunraysia (western NSW and Vic) and SA- Riverland showed that the average soil salinity under the drip was always higher than the sprinkler. Two important processes for salt accumulation under the drip are: a non-uniform irrigation application and associated salt leaching efficiency of the profile.

For grapes, the threshold salinity expressed in saturated soil paste extract EC (ECe) is 1.5 dS/m (Maas and Hoffman 1977). The threshold is the maximum ECe value that a crop can tolerate without a potential yield decline. The salinity tolerance for grapevines was reassessed under Australian conditions (Walker and Stevens 2004), where for an own rooted vine the value was raised to 1.8 dS/m. From eqn (13), the SWE EC (ECsw) is twice the ECe therefore, threshold ECsw for own rooted grapes should be 3.6 dS/m. Examining the seasonal variability of ECsw data the average root zone salinities present no risk of yield loss. However, high water use efficiency coupled with poor quality irrigation water and a low efficiency of salt leaching will necessitate the use of SWE or some other form of root zone salinity monitoring in order to prevent major salinity induced yield loss and wine quality deterioration.

Grape's salt tolerance needs critical evaluation given that in most cases it is not yield but quality that is the criterion and hence typical Maas & Hoffman threshold values may not valid anymore. A new criterion needs to be established based on quality, eg Cl and Na content of fruit juice (European Union limits for Cl & Na in wine are: 600 and 394 respectively). Secondly, the salinity impact varies with the EC during the season; probably more importantly, the impact depends on at what crop physiological stages the salinity spike(s) occur.

7. Drainage and Reuse of drainage water

7.1 Subsurface Drainage

Subsurface drainage has been used extensively in arid and semi-arid irrigated agriculture for the control of waterlogging and salinisation. Research has shown that two main factors affect drainage water salinity (Fio and Deverel, 1991; Grismer 1993; Guitjens et al., 1997; Christen and Skehan, 2001; Ghaemi and Willardson, 1992; Hornbuckle et al,. 2005a):

1. Depth and spacing of the drains, which largely influence the water flow paths to the drains

2. Soil salinity and its distribution with depth.

Previous research (Ayars et al., 1997; Fio and Deverel, 1991; Guitjens et al., 1997; Jury, 1975a) has shown that drainage design and management can have a large effect on the quantity of subsurface drainage water and subsequently the salt loads generated. This is due to water flow paths to drains being controlled by a number of variables, such as drain depth and spacing as well as irrigation management.

In a review of subsurface drainage systems (Christen et al., 2001) in irrigation areas in Australia, it was shown that in many cases the drainage salt loads are often 5-10 times greater than that applied through the irrigation water, even after the reclamation phase was completed. Thus indicating these drainage systems typically remove stored salt as well as that applied with the irrigation water, (Figure 11). Often this stored salt may originate from below the root zone with its removal offering little benefit to the crop.

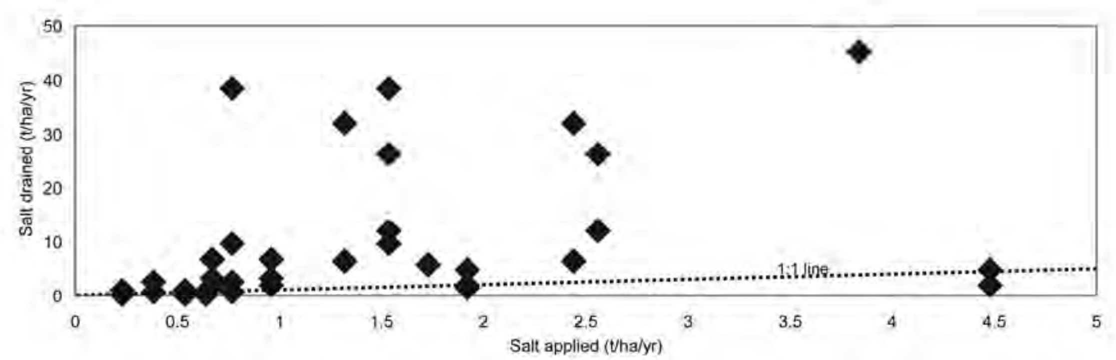

Figure 11. Salt applied and salt drained based on studies undertaken in Australia in 10 irrigation areas (Christen, et al., 2001)

From a study of information provided in reviews and studies of irrigated salinity (Blackwell et al., 2000; Evans, 1989; Hillel, 2000; Tanji, 1990; Westcot, 1988) the main control and management options of subsurface drainage water are depicted in Figure 12. Drainage water initiates from leaching of agricultural fields through the removal of excess water applied to the field either through irrigation or rainfall. Volumes of water leaving the system are controlled by management variables shown in Figure 12.

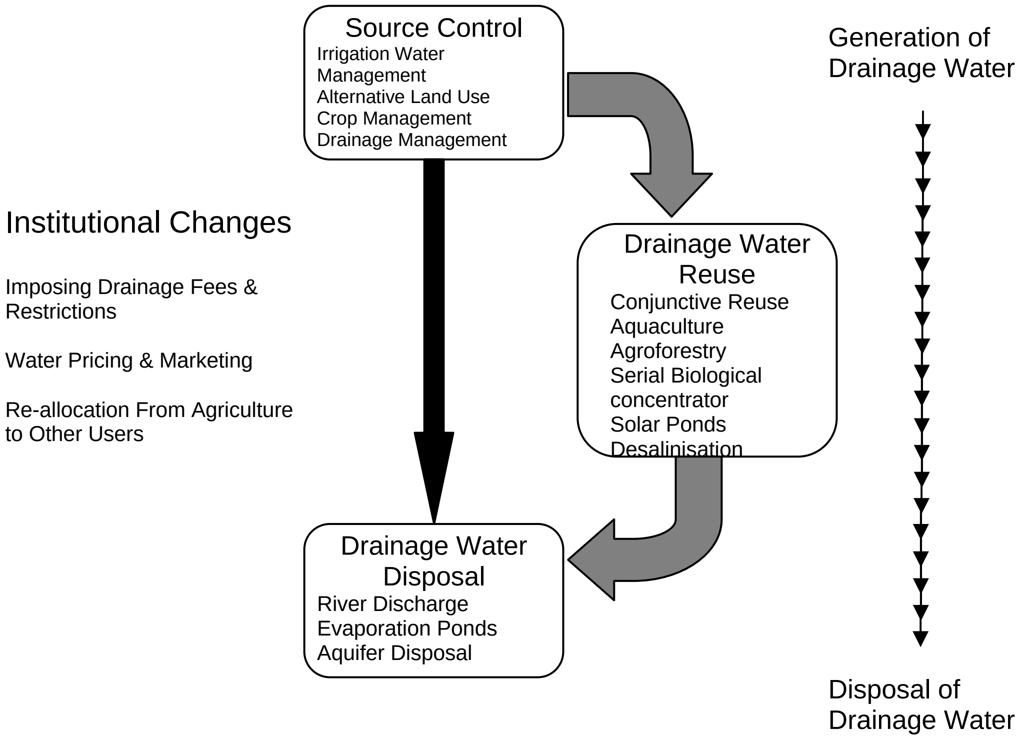

Figure 12. Options available for reducing subsurface drainage water from irrigated agriculture.

Four main options exist for managing saline subsurface drainage water and these are institutional changes, drainage efficiency improvement, drainage water reuse and drainage water disposal. While each of these options is important and may play a vital role in sustaining irrigation, central to all options is drainage efficiency improvement. Many of the other options rely heavily on drainage efficiency improvement to provide a manageable volume of drainage water that needs to be disposed for the economics of such systems to be favourable (Blackwell et al. 2000; Evans 1989).

7.1.1 Subsurface Drainage Performance Assessment

Subsurface drainage provides a valuable method of reclaiming salinised land for agriculture and managing salinity. However, rarely is any form of performance assessment been undertaken (Christen et al., 2001). Performance assessment should be undertaken to determine management of salinity in the rootzone and spatial variation and to assess the opportunity for implementation of controlled drainage to provide irrigation water savings and minimise downstream impacts (Hornbuckle et al., 2005a). One approach to this problem is to use an electromagnetic survey, such as EM38, to ascertain the spatial distribution of root zone soil salinity. A calibrated survey can provide a quantitative estimate of the reduction in soil salinity and hence the effectiveness of the drainage system. Calibration of EM survey can be optimised by processing the data from the survey using the ESAP (ECe Sampling, Assessment and Prediction) software from the U.S. Salinity Laboratory, California (Lesch, et al. 2000; website http://www.ussl.ars.usda.gov/models/esap-95.htm).

The calibrated data can then be used to map soil salinity variation over time to ascertain the uniformity of soil reclamation and when drainage management may commence (Figure 13).

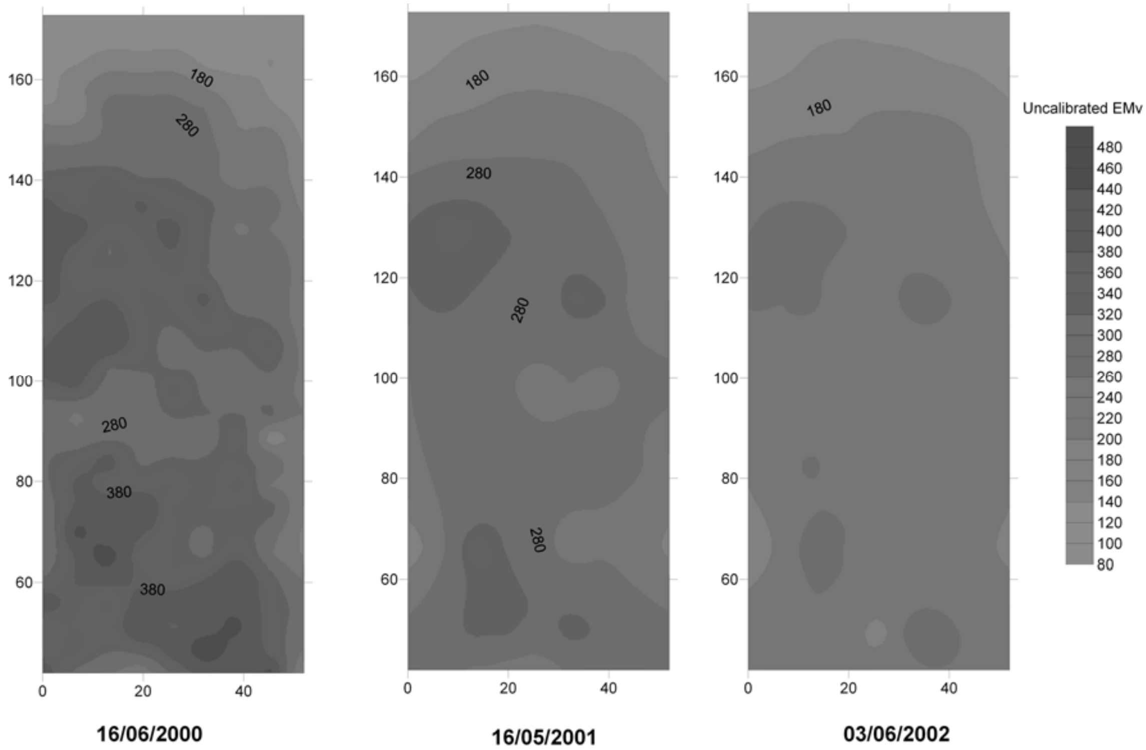

Figure 13. EM38 survey showing soil reclamation after drainage (Christen et al., 2004).

The soil survey data can also be used to predict relative yield reduction and hence financial impact (Figure 14). This can be used to access whether drainage to control salinity is likely to be economically beneficial.

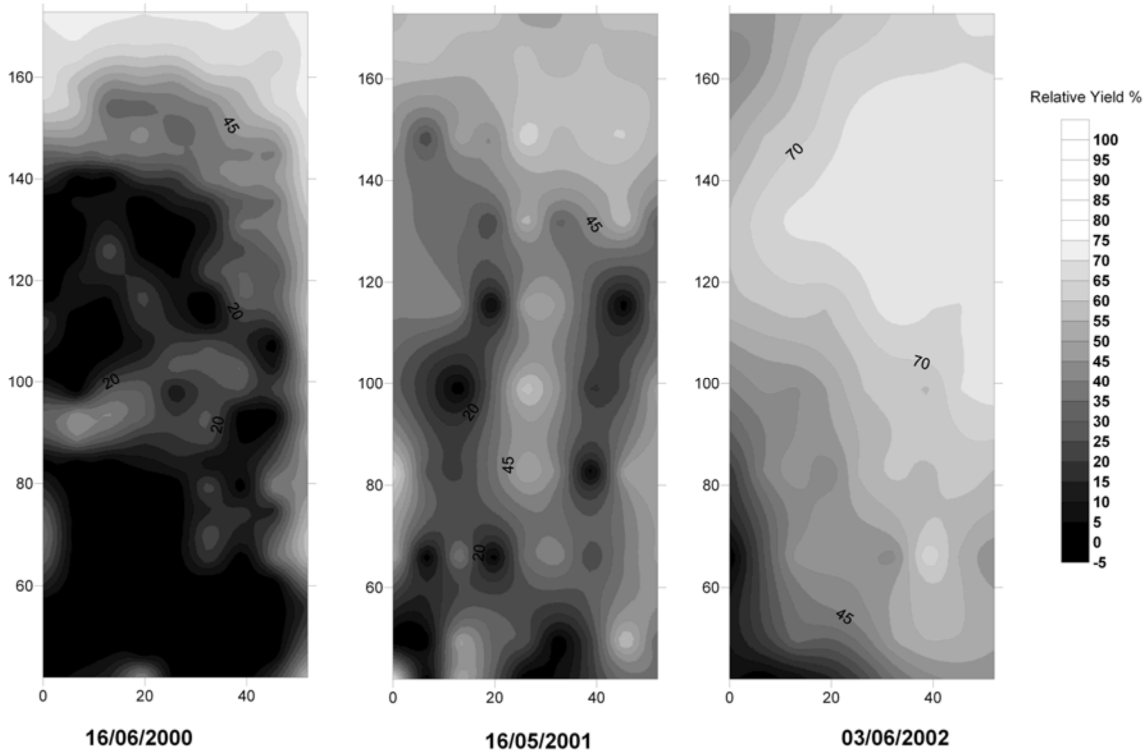

Figure 14. Relative yield for grapevines with soil salinity distribution (Christen et al., 2004).

The procedures and examples for using these approaches for drainage design and management to minimise downstream impacts and conserve irrigation water can be found in Christen et al. (2004).

7.2 Regional drainage

The implications of management on and interventions to existing drainage systems are complex and often have the potential to cause significant impacts on stakeholders in the system unless careful consideration is given to all aspects of the system. There is a need to understand clearly the tradeoffs between management options and interventions, and their impacts on drainage return flow and water quality. This requires water accounting (quantity and quality) to be conducted at the regional scale. Tools or frameworks which allow all aspects of the drainage intervention to be considered and trade-offs between stakeholders to be investigated, will allow improved decision making. When such accounting has not been done it has often led to a further different set of problems, often transferring the problem downstream.

At the regional scale tools for managing drainage and solutes are limited. Some detailed numerical studies using Modflow/MT3DMS based models have been undertaken in the Western US (Gates et al., 2005; Burkhalter and Gates, 2006; Schoups et al., 2005). These models require considerable inputs and expertise to be used effectively and have mostly focused on groundwater issues. Recently (Hornbuckle et al. 2005b) developed another approach using a 1-D soil water balance model for representing irrigated land uses coupled to a nodal network model to specifically look at return flows from irrigation areas to rivers systems which are generated from drainage (Figure 15). Paydar et al. (2005) have also recently developed an approach of coupling recharge generated from a 1-D soil water balance model to a regional groundwater model.

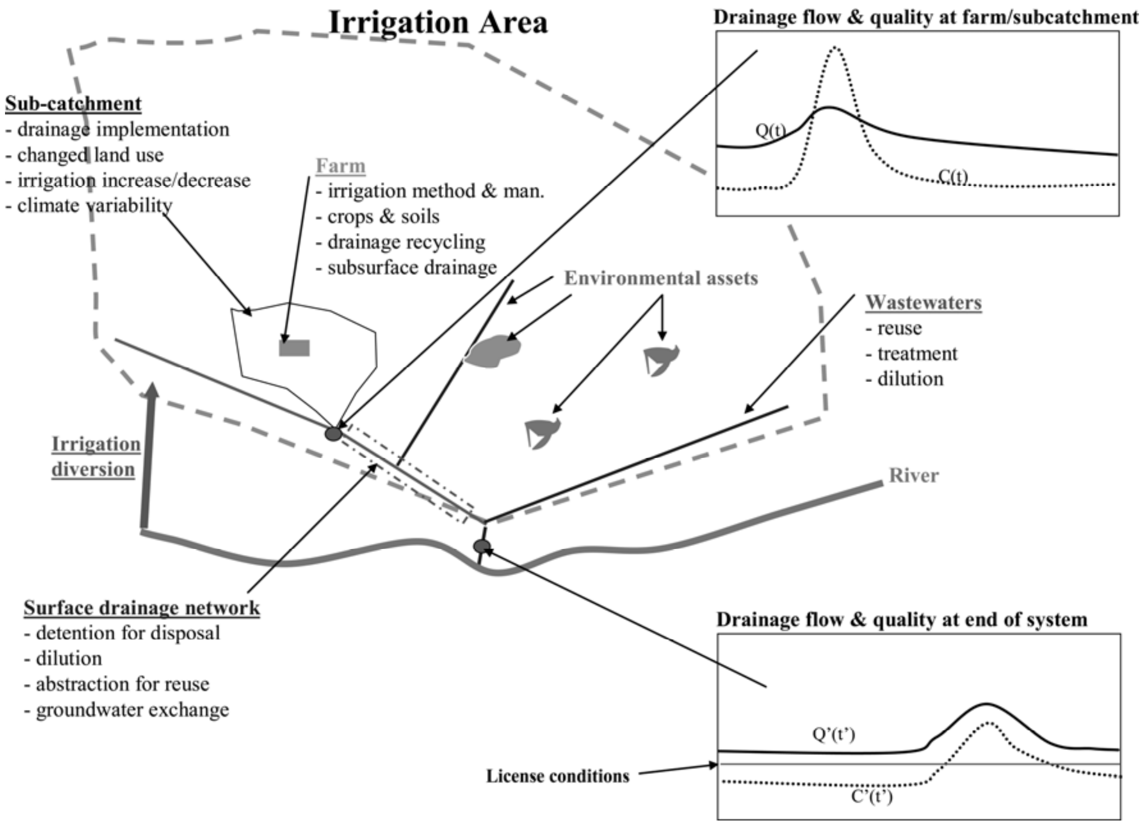

Figure 15. Conceptual model of drivers and management levers dictating quality and quantity of irrigation return flows (after Hornbuckle et al., 2005).

7.3 Drainage water reuse options

The drainage water reuse management options are discussed in detail in Tanji and Kielen (2002) and relevant extracts from this review publication are reproduced below. These options are illustrated in Figure 16 (Tanji and Kielen, 2002).

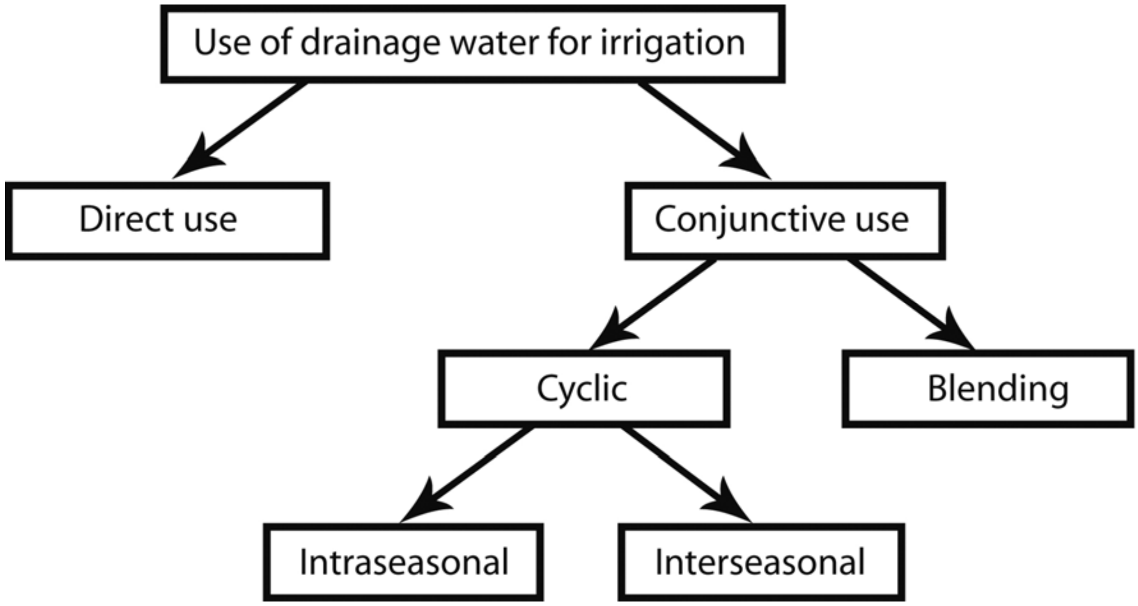

Figure 16. Use of drainage water for crop production. Source: Tanji and Kielen (2002)

Drainage water of sufficiently good quality can be used directly for crop production. Otherwise, drainage water can be reused in conjunction with fresh water resources (Figure 16). Conjunctive use involves blending drainage water with fresh water. Alternatively, drainage water can be used cyclically with freshwater being applied separately. In cyclic use, the two water sources can be rotated within the cropping season (intra-seasonal cyclic use), or the two water resources can be used separately over the seasons for different crops (inter-seasonal cyclic use). The choice of a certain reuse option depends largely on factors such as drainage water quality, crop tolerance to salinity and availability of fresh water resources. The quantity and time of availability of drainage water is of major importance. For example, where reuse takes place in an irrigation system in which fresh water is only available sporadically then the probable mode of reuse is limited to either direct or cyclic use.

7.3.1 Direct use

The direct use of drainage water is implemented mainly at the farm level, whereby the drainage water is not mixed with freshwater resources. Research results from India, Pakistan, Central Asia and Egypt, where surface irrigation methods are applied, show that drainage water can be used directly for irrigation purposes without severe crop yield reductions where the salinity of the drainage water does not exceed the threshold salinity value for the crops grown and good drainage conditions exist. As crops are often more sensitive to salinity during the initial growth stages, research in India has revealed the importance of pre-irrigation with good quality irrigation water. Higher crop yields were attained when freshwater pre-irrigation was applied with only drainage water being applied thereafter. Under these conditions, drainage water with salinity levels exceeding the threshold value could be used while maintaining acceptable crop yields. The long-term sustainability of direct use of drainage water depends on maintaining a favourable salt balance and preventing soil degradation due to sodicity problems.

7.3.2 Conjunctive use - blending

Where drainage water salinity exceeds the threshold values for optimal crop production, it can be mixed with other water resources to create a mixture of acceptable quality for the prevailing cropping patterns. In the Shepparton irrigation area saline groundwater pumped for irrigation is blended with surface irrigation water to below 0.8 dS/m before applying to the clover based pasture grown on the farms, based on field research findings.

7.3.3 Conjunctive use - cyclic use

Cyclic use, also known as sequential application or rotational mode, is a technique that facilitates the conjunctive use of freshwater and saline drainage effluent. In this mode, saline drainage water replaces canal water in a predetermined sequence or cycle. Cyclic use is an option where the salinity of the drainage water exceeds the salinity threshold value of the desired crop. A condition for cyclic use is that two different water sources can be applied to the field separately. Modelling and field studies have demonstrated the feasibility of the cyclic reuse strategy (Rhoades, 1987; Rhoades *et al*., 1988a,b; Rhoades *et al*., 1989).

The cyclic use of drainage water can be either intraseasonal or interseasonal. The latter mode of cyclic use follows the same principles for each cropping season as the direct use of drainage water. Cyclic use also requires attention to soil degradation as a result of using sodic water.

7.3.4 Reuse in specialised Integrated Farm Drainage Management (IFDM) systems

In addition to the drainage water reuse management options illustrated in Figure 16, specialised system that could be developed to address the site-specific local situations are described in detail in Tanji and Kielen (2002). One of these, the IFDM systems, aims to utilise drainage water as a resource to produce marketable crops and to reduce the volume of drainage water to be discharged (SJVDIP, 1999; and Cervinka *et al*., 2001).

Under IFDM, drainage water is used sequentially to irrigate crops, trees and halophytes with progressively increasing salt tolerance. Each time the drainage water is reused, the volume of effluent is reduced and the salinity concentration increased. A typical IFDM system consists of four zones. In Zone 1, traditional salt sensitive crops are grown, e.g. vegetables, fruits, beans and corn. In Zone 2, traditional salt tolerant crops are grown, e.g. cotton, sorghum and wheat. In Zone 3, salt tolerant trees and shrubs are grown. In Zone 4, only halophytes can be planted. The final non-reusable drainage water is discharged in a solar evaporator.

The solar evaporator consists of a levelled area lined with plastic on which the brine is disposed and the crystallised salts are collected. The daily discharge of drainage water corresponds to the daily evaporation, to prevent water ponding that attracts waterbirds. This is only important where high concentrations of toxic trace elements are present in the drainage water, otherwise a normal evaporation basin can be used.

In Australia, an IFDM system referred to as the Sequential Biological Concentration (SBC) system has been developed to manage saline surface and subsurface drainage waters in irrigation areas (Blackwell *et al.*, 1999, Mann *et al.*, 2003), and successfully demonstrated in trial plots.

7.3.5 Other drainage water reuse management and disposal options

In addition to agricultural reuse, drainage waters could be used or disposed through other methods. Saline drainage waters could be utilised in reclaiming sodic soils. Sodic soils often have low hydraulic conductivity with low saline irrigation waters as a result of the high sodium percentage on the soil exchange complex. The reclamation of sodic soils requires that a divalent solute (mainly calcium) pass through the soil profile, replacing exchangeable sodium and leaching the desorbed sodium ions from the rootzone. Therefore, the rate at which sodic soils can be reclaimed depends on the water flow through the soil and the calcium concentration of the soil solution. The application of leaching water with a high electrolyte concentration promotes flocculation of the soils and thus improves soil permeability. This expedites the reclamation process. Amendments need to be added to replace sodium with calcium ions on the soil exchange complex. Over time, less-saline water needs to replace the saline leaching water to lower the salinity levels sufficiently to establish salt sensitive crops. The use of saline drainage water to reclaim salt-affected soils is not a permanent solution for reducing drainage effluent disposal volumes. It is only a substitute for the use of good quality irrigation water for reclamation purposes. Abrol et al. (1988) has compiled a list of procedures and measures for the reclamation and management of saline and sodic soils. Drainage waters could also be reused in wildlife habitats and wetlands or disposed to on-farm or communal evaporation ponds.

7.4 Australian field research studies on reuse of subsurface drainage and other poor-quality wastewaters

The long-term sustainability of reusing subsurface drainage water and similar poor-quality wastewater has been evaluated in field trials at several sites (Surapenani and Olsson 2001, Blackwell *et al.* 1999, Mann *et al.* 2003, Stevens *et al.* 2003, Jayawardane *et al.* 2004).

7.4.1 Conjunctive water use (CWU) schemes in Shepparton Irrigation Region in northern Victoria

The Land and Water Salinity Management Plan of Shepparton Irrigation Region (SIR), promotes groundwater pumping and re-use for irrigation where groundwater quality and availability allow dilution with channel water, termed ‘Conjunctive Water Use’ (CWU), to levels that produce minimal production losses from annual and perennial pastures used widely for dairying. In addition, municipal and industrial wastewaters are used on a smaller scale for irrigating pastures and some crops.

In a field assessment of the impact of this CWU strategy on the irrigation region, Surapaneni and Olsson (2001) reached the following conclusions in their review paper. “Although the strategy has, so far, achieved acceptable control of soil salinity levels in the crop root zones, while generally maintaining pasture yields, a concern that ‘conjunctive water use’ may not be sustainable in the long term arises from the sodicity of the groundwater and wastewaters. The continual addition of sodium to clay soils, initially low in both sodium and electrolytes (upper 0.5 m depth), risks the soils becoming sodified, with attendant soil physical problems should salts be leached to below threshold electrolyte concentrations, as in winter for example.”

7.4.2 Serial Biological Concentration system in Northern Victoria

Mann *et al.* (2003) also expressed similar concerns on the observed slow and steady rises in groundwater salinity in hot-spots within SIR under CWU operations. An area of about 70,000 ha in SIR has been currently identified as unsuitable for the CWU system.

The sequential biological concentration technique has been suggested as an alternative approach. A trial SBC project has been carried out to reuse the drainage water pumped from a shallow saline aquifer on a 3-ha tile-drained forestry cropping area, to reduce the volume of water for disposal in an evaporation pond. Salt balance was achieved in the forestry area of the SBC pilot trial over the 4-year observation period. In the evaporation basins, out of 524 tonnes of salt discharged into the ponds, only 125 tonnes could be accounted for in the water stored in the ponds after 4 years of operation, indicating considerable leakage.

7.4.3 Reclaimed water (RCW) reuse on the Northern Adelaide Plains (NAP) horticultural districts

Reclaimed water (RCW) reuse has been practiced on the Northern Adelaide Plains (NAP) horticultural districts for more than 28 years. The RCW has had approximately 1.7 times the salinity and twice the sodium absorption ratio (SAR) of bore water commonly used for irrigation in the district. Recently, a large-scale reclamation scheme has been commissioned which could eventually supply approximately 30 GL of RCW to over 250 growers on the NAP.

Stevens *et al.* (2003) studied the effects of reclaimed water reuse on the NAP. Their study compared historical water quality and time of use data with physio-chemical properties of soil cores taken from sites where reclaimed (RCW-irrigated) or bore water had been used for irrigation, or sites that had not been irrigated (virgin). The aim was to determine if current farming practices irrigating with RCW could, now or in the future, lead to a decrease in yields through detrimental increases in soil salinity, sodicity, and boron (B) concentrations, and to determine if these changes were significantly different from bore-irrigated or virgin sites. Stevens et al. (2003) summarised the results of the study as follows. “Data suggested that changes in soil salinity and B concentration from RCW use would not decrease yields. However, changes in soil SAR had the potential to restrict drainage and consequently increase salinity; although a more functional critical SAR value for the NAP soils needs to be defined to assess this potential. These findings suggest that farming methods, in the 1967-95 period, did not address the physico-chemical changes associated with the use of more sodic RCW. Considering the future scale of RCW use, the SAR of the irrigation water may need to be decreased and/or appropriate farming methods developed and practised with the use of RCW to protect these soils for future horticultural activities”.

Their data also showed that although the soil salinity in the shallow rooting depths of vegetable crops was below critical limits, salt accumulation was observed at lower depths which could affect cropping with deeper rooted crops. There was insufficient winter rainfall to leach the salt from these deeper soil layers.

7.4.4 The Land FILTER system to renovate saline sewage effluent in Griffith

Studies on use of saline and sodic wastewaters on soils with restricted drainage have indicated development of salinity and sodicity problems. Falkiner and Smith (1997) found that 4 years of irrigation with slight to moderately saline and sodic effluent resulted in marked increases in soil salinity and the increase in soil sodicity to around

20-25%. Smith *et al.* (1996) showed that excess water needed to be applied at this effluent irrigated plantation to promote the leaching of excess salt accumulating in the root zone of the soil profile.

The FILTER (Filtration and Irrigated Cropping for Land Treatment and Effluent Reuse) technique was developed as a new controlled flow system for sustainable land treatment and reuse of poor-quality polluted and saline wastewaters on soils with restricted drainage (Jayawardane 1995, Jayawardane *et al.* 1997a,b, 2004). The FILTER technique combines the use of nutrient-rich wastewater for intensive cropping, with filtration through the soil to a subsurface drainage system, Figure 17. Wastewater application and subsurface drainage in the FILTER system are regulated to ensure adequate removal of pollutants, thereby producing minimum-pollutant drainage water which can meet the stringent Environmental Protection Authority (EPA) criteria for discharge to sensitive inland surface water-bodies. The use of the subsurface drainage system to remove the excess water during the periods of low cropping activity or periods of high rainfall allows the use of this technique throughout the year without the need for costly wet weather storages on urban lands.

Preliminary testing of the FILTER technique was carried out on a heavy clay soil with impeded drainage at the Griffith sewage works site, on one-hectare plots. This was followed by field evaluation of a 15-hectare pilot FILTER system. The field data showed that the FILTER system met its objectives of reducing nutrients and other pollutants in the drainage waters below EPA limits for sensitive waters, while maintaining adequate drainage flow rates. Significant crop yields and nutrient removal were obtained, which would help to maintain nutrient balance required for a sustainable system, and to offset costs in a commercial system. The other beneficial effects were reduced suspended solids and E.coli, and an increased N: P ratio in the drainage waters.

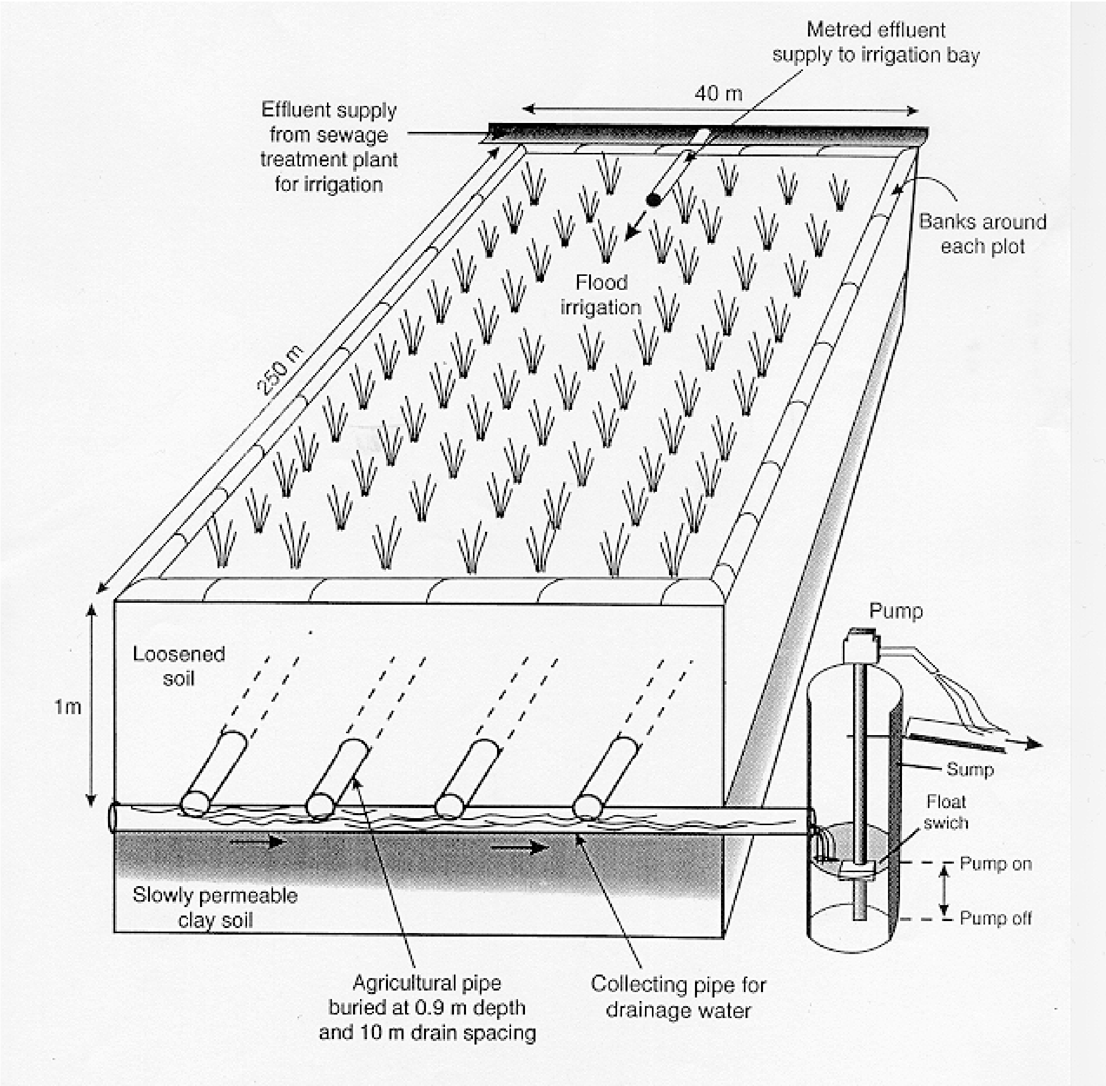

Figure 17. Diagram of the land FILTER system's engineering layout.

The use of the FILTER system on the highly saline-sodic soil resulted in a progressive decline in salinity and sodicity. The concentration of salt increased in the drainage waters, mainly due to leaching of salts that had accumulated in the soil through previous effluent application without sub-surface drainage, as well as salt concentration by evapotranspiration. After salt equilibration is reached through leaching of these accumulated salts, the salt load in the drainage water will be the same as in the effluent, while the salt concentration changes will depend on the balance between concentration through evapotranspiration and dilution through rainfall.

Thus, the FILTER technique can provide a potentially sustainable system to treat different polluted wastewaters with suitable design and management to meet the specific wastewater site conditions and requirements.

7.4.5 Sequential biological concentration system to manage saline drainage waters in Murrumbidgee Irrigation area

The use of the sequential biological concentration (SBC) technique was proposed (Blackwell *et al.* 1999) as a community based solution to the problems of managing drainage wastewater from the Murrumbidgee Irrigation area (MIA) which contains a cocktail of pollutants and salts. SBC technique is based on using a modified FILTER system. Previous field studies on the FILTER system indicated that it was able to remove all pollutants in MIA drainage waters, except the salts, which pass through unabsorbed by the soil in the FILTER plots. Therefore, in conceptually developing the SBC system, the FILTER system had to be modified to economically manage these salts through a process of water re-utilisation, salt concentration and eventual salt removal. The results of the field trials on the SBC system are summarised by Blackwell *et al.* (1999).

8. Summary

Here we have presented a review of knowledge on the ‘The State of Measuring, Diagnosing, Ameliorating and Managing of Solute effects in Irrigated Systems’. We briefly discussed the dimensionality of systems as a way to define the ‘state space’ of the system which will allow a structure for modelling and analysis.

The effects of salinity and sodicity on soil properties were presented and the threshold electrolyte concentration (TEC) used as a concept for management of irrigation. The effect of crop growth in relation to specific solutes and the salt load was presented and state of the art in managing crop growth, with saline water and leaching fractions has been around a long time. Recent evidence that leaching efficiency is not 100% would suggest that this concept needs revision.

Models for water and solute movement were discussed and we suggested that some of the analytical models have not had the attention they deserve. This review does not provide a comparison or extensive list of models but does present references to models which are the present state of the art.

A review of information on amelioration of saline and sodic soils shows that the introduction of 2-D irrigation systems and a desire to increase water use efficiency will need to be tempered by the potential for salinisation of soils. There will be a need to calculate how much salt can be accumulated in the profile and over what time period.

Recent reviews of electromagnetic methods for measuring electrical conductivity are referenced and these give a comprehensive view of this subject. Here we have presented some of the difficulties these reviews suggest are associated with measuring electrical conductivity using electromagnetic and other methods. These show that the use of such methods without calibration is fraught with possible misinterpretation of the results.

Methods and models for assessing drainage at regional scales are introduced. The reuse of drainage water is review with emphasis given to a number of methods that have been introduced both in Australia and internationally. All of these options require good monitoring of the soil drainage system if adoption is to occur. They do, however, allow for the otherwise ‘waste’ nutrients to be utilised.

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- NOTES -

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