THE POTENTIAL FOR SOIL SALINIZATION ABOVE AQUIFERS INFLUENCED BY SEAWATER INTRUSION

A.D. Werner[A] *and D.A. Lockington*[B]
[A] Department of Natural Resources, Mines and Energy, Brisbane, Australia.
[B] Department of Civil Engineering, University of Queensland, Brisbane, Australia.

Abstract
In coastal areas of Queensland, seawater intrusion has lead to salinization of the groundwater resources. Seawater intrusion in shallow alluvial aquifers similar to those found near the Queensland coastline may lead to salinization of the water table. In certain circumstances, the soil above such a water table may become salt damaged due to the transport and concentration of saline water under capillary rise and evapotranspiration. In this paper, we explore the influence on the potential for soil salinization of a range of factors, including water table salinity, soil type, depth to the water table and potential evaporation. The relationship between water table salinity changes induced by seawater intrusion, and the potential for soil and ground surface salinization to occur, is examined through numerical modelling. Adopting a sinusoidal forcing pressure wave at the lower boundary of the numerical model represents fluctuations in the water table due to tidal impacts.

The results of numerical experiments indicate that soil salinization occurs more rapidly with increasing water table salinity. The occurrence of salt accumulation at the ground surface was relatively insensitive to water table salinity, although once initiated, the rate of salt accumulation was shown to be strongly dependent on the water table salinity.

Additional Keywords: soil, salinization, seawater intrusion.

Introduction
In Australia, the occurrence of *dryland salinity* has caused considerable damage both to the ecosystem and to arable land. Losses in agricultural production and infrastructure are estimated as $230 million per annum for 2003 (CSIRO Land and Water, 2003). To assess the potential for increasing dryland salinity in Queensland (and the Murray-Darling basin), the Queensland Government has undertaken *salinity hazard mapping* to identify high-risk soil salinization sites (Gordon et al., 2002). These maps indicate a considerable risk of soil salinization at several locations near the Queensland coastline. In some of these coastal areas (eg. Burdekin Delta and Bundaberg), the aquifers have experienced or are considered susceptible to seawater intrusion.

The predominant precursor to dryland salinity and similar soil salinization phenomenon (eg. irrigation and urban salinity) is an increase in water table elevations resulting from changes to the hydrologic cycle, namely increases in groundwater recharge (Pavelic et al., 1997). The rise in groundwater levels may induce salinization of the vadose zone or an accumulation of salts on the ground surface, initially where the terrain is low lying, through a combination of capillary rise, transpiration and evaporation. Any rise in water table heights may also mobilise previously suspended vadose zone salts, further exacerbating the build-up of soil salts. Where groundwater discharges to surface water bodies, elevated water table heights increase groundwater discharge rates, producing an adverse impact on fresh surface water systems where the groundwater quality is poor.

While studies of dryland salinity highlight the importance of rising water tables (eg. Cook et al., 2001), no study of the influence of water table salinity on soil salinization, particularly in relation to near-coastal aquifers susceptible to saltwater intrusion is apparent in the literature. In this study, we examine the changing potential for soil salinization with variation to the water table salinity. Increases in water table salinity are inferred from two-dimensional hypothetical simulations of saltwater intrusion in a coastal alluvial aquifer. The importance of water table salinity is compared to such factors as water table depth from ground surface, soil hydraulic properties and tidal forcing of the groundwater system.

Seawater Intrusion and Water Table Salinization
The potential for seawater intrusion to increase the groundwater salinity at the water table has been investigated in a separate study by Werner (2004), in which an estuary-aquifer system was explored using two-dimensional numerical modelling and field observations of groundwater salinity. The study primarily focused on the Pioneer Valley, which is characterised by macro-tidal estuaries and a shallow alluvial aquifer system. Pioneer Valley

estuaries transport marine salts up to 11 km inland, and groundwater pumping near these estuaries has induced a lateral movement of the freshwater-saltwater interface in the adjacent groundwater system. Pumping restrictions have been enforced since 1993 in an attempt to reduce the landward encroachment of this estuarine seawater intrusion. However, field observations of groundwater electrical conductivity (EC) in near-estuary observation bores and site-specific numerical modelling (Werner, 2004) indicate that a potential exists for estuarine seawater intrusion to continue increasing in some areas.

While seawater intrusion commonly propagates preferentially along the aquifer basement due to density effects, in shallow alluvial aquifers similar to those found near the Queensland coastline, seawater intrusion may also lead to salinization of the water table (Werner, 2004). Under certain circumstances, the soil above such a water table may become salt damaged due to the transport and concentration of saline water under capillary rise and evapotranspiration. In this paper, we explore the influence on the potential for soil salinization of a range of factors, including water table salinity, soil type, depth to the water table and potential evaporation.

Experimental Setting

The potential for vadose zone salinization is explored using numerical experiments of variably saturated flow and solute transport. The modelling code HYDRUS-1D (Simunek et al., 1998) is adopted. In brief, HYDRUS-1D is a one-dimensional simulator of water, solute and/or heat movement in variably saturated media. Unfortunately, HYDRUS-1D is unable to account for density-dependent flow and as such, this paper is restricted to density-independent water and salt transport predictions. Ignoring density-effects is expected to over-predict the upward movement of salts.

The analysis of vadose zone salinization presented here considers a selection of the primary processes, which are represented in mostly hypothetical numerical experiments. Simulation parameters are assumed to approximate an unconfined alluvial aquifer, typified by that encountered in the Pioneer Valley, northern Queensland. While unsaturated soil heterogeneities are expected to affect the vadose zone movement of solutes over a range of different conditions (eg. van Dam and Feddes, 2000), the simulations presented here are restricted to homogeneous materials. A fabrication of vadose zone heterogeneities and subsequent analyses, in the absence of field data, is considered beyond the scope of the current study.

Three soil types are adopted: sand, sandy clay loam and sandy clay, with hydraulic properties taken from Carsel and Parrish (1988). Although seawater intrusion experiments by Werner (2004) indicate that water table salinization in sandy clay loam is extremely slow, it is possible that a sandy clay loam overlies a more permeable aquifer material, such as sand. The upward-fining sequences encountered in Pioneer Valley lithological descriptions (Murphy and Sorensen, 2000) support this hypothesis. Hence, two fine-grained materials (sandy clay loam and sandy clay) and one coarse-grained material (sand) are considered in the numerical experiments.

Evaporative losses at the ground surface are expected to drive soil salinization (Cramer and Hobbs, 2002) and as such, an evaporative boundary condition is adopted at the upper model limit. While it is understood that transpiration losses will contribute to the process of vadose zone transport, surface evaporation from bare soil and evapotranspiration are assumed equivalent for the purposes of this study. Cook and Rassam (2002) also adopted this assumption in an investigation of high water table drainage.

In all simulations, a fluctuating (sinusoidal) water table is adopted at the lower boundary condition as a representation of tidal fluctuations (Figure 1). Tidal boundary conditions in vadose zone modelling are not meant to reproduce the water table oscillations in the field (i.e. in the Pioneer Valley), but are included merely to provide an arbitrary response to the nearby estuarine tidal forcing, with wave amplitude and period approximating possible groundwater wave values (i.e. 1 m and 13 hours, respectively).

Results and Discussion

The sequence of numerical experiments reflects the objectives of the study. The potential for vadose zone salt build-up was explored in salt transport simulations of constant evaporation. Various average water table depths from the ground surface, forcing wave amplitudes, water table relative salinities and soil types were examined.

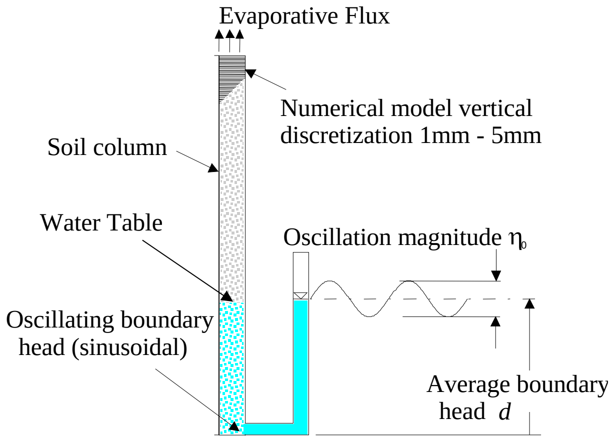

Figure 1. The layout of 1D vadose zone numerical experiments.

Water Table Depth
The first simulation was based on typical Pioneer Valley conditions (north of Sandy Creek), whereby the mean water table depth was 3 m below the ground surface. The hydraulic properties of *sandy clay* (Carsel and Parrish, 1988) were adopted. The forcing pressure wave at the lower boundary had a period of 13 hours and amplitude of 1 m, which induced a water table wave of amplitude approximately 145 mm.

A salinity of 1.0 (relative to seawater) was adopted at the lower boundary to indicate the *worst-case* water table conditions. The initial relative salinities below and above the water table were taken as 0.95 and 0, respectively. The solute dispersivity α_L was taken to be 0.2 m. Estimates of α_L in similar systems are given by Grifoll and Cohen (1996) and the value adopted here approximates the median of literature values. Upstream weighting in the iterative solution of salt transport was adopted to reduce numerical oscillations (despite low discretization parameters – the grid Peclet number and the Courant number), and hence some artificial numerical dispersion was unavoidable (p146, Zheng and Bennet, 2002). The influence of adopting a lower dispersivity is described briefly below.

The potential evaporation at the ground surface was taken as the average annual pan evaporation for Mackay of 5.5 mm/d multiplied by a climatic factor of 0.7 (DPI, 1994). In HYDRUS-1D, the evaporative flux at the ground surface reduces with the soil matric potential (and the unsaturated hydraulic conductivity) until a specified pressure head h_A is reached, upon which the surface boundary is defined by a Dirichlet condition at head h_A (Simunek et al., 1998). In the current simulations, h_A was arbitrarily selected as –40 m. The sensitivity of the value h_A on transport predictions is also described below.

The predicted salt migration into the vadose zone is illustrated in Figure 2 for the *sandy clay* soil. Clearly, the predicted salt movement was minimal in this system. The evaporative flux at the surface reduced quickly, and the specified minimum surface pressure head h_A of –40 m was reached after 90 minutes. The evaporative flux at the surface reduced from the initial value of 3.9 mm/d to 0.064 mm/d after 29 days, and to 0.019 mm/d after 1 year. The low saturated hydraulic conductivity of the sandy clay (i.e. 0.029 m/d) combined with the predicted soil saturation of 34% at the ground surface after 1 year resulted in an unsaturated hydraulic conductivity here of 5.8×10^{-9} m/d. Hence, the potential for salt build-up was restricted by the depth to the water table (and thus the low soil saturation) and the saturated hydraulic conductivity of sandy clay.

Clearly, the water table depth of 3 m was sufficient to minimize vadose zone salinization in the hypothetical system described above (sandy clay) in the time frame considered (1 year). Simulations conducted using alternative soils (results not shown for brevity) also produced only limited vadose zone salinization during the prediction period. Alternative soils had either lower saturated conductivities, which reduced salt flux above the water table, or lower surface saturations, which limited evaporative fluxes. For example, Carsel and Parrish (1988) *sand* at –40 m pressure head had an unsaturated hydraulic conductivity of 3.5×10^{-32} m/d, although the saturated hydraulic conductivity was 7.1 m/d. The Carsel and Parrish (1988) *silty clay* had an unsaturated hydraulic conductivity of 4.6×10^{-8} m/d at -40 m pressure head, while the saturated conductivity was 0.0048 m/d.

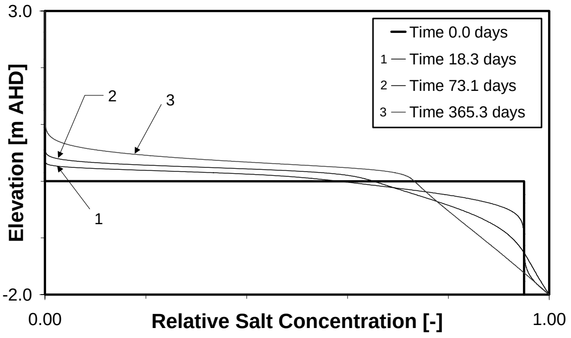

Figure 2. Changes to the vadose zone salinity during evaporation from sandy clay.

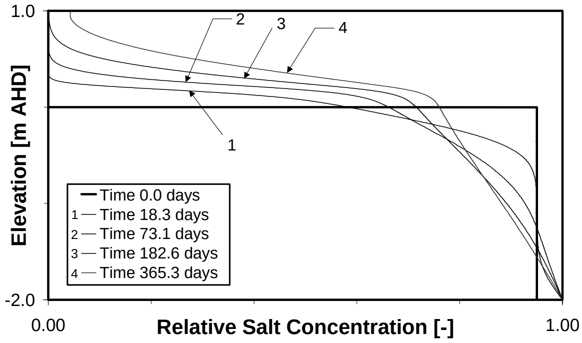

Figure 3. Vadose zone salinity changes for an average water table depth of 1 m.

The results described above indicate that a water table depth of 3m was sufficient to limit ground surface salinization in all soil types during a 1-year period of continuous evaporation, where the soil profile was initially non-salty. While the 3 m water table depth was indicative of some parts of the Pioneer Valley near Sandy Creek, water table depths were significantly less in other near-estuary regions, in particular on the southern side of Sandy Creek. There the soils range from loamy sand to clay. Hence, two addition simulations were conducted using average water table depths of 2 m and 1 m to ascertain the effect of water table depth on the potential for vadose zone salt build-up. The hydraulic parameters of *sandy clay* (Carsel and Parish, 1988) were adopted.

Simulated vadose zone salinities from water table depths of both 3 m and 2 m were essentially the same. However, with the water table at 1 m below the ground surface, salt accumulation at the soil surface was predicted to occur (Figure 3). Surface salt concentrations of 0.001, 0.01 and 0.045 were obtained after 187, 265 and 365 days, respectively. The 1-year evaporative fluxes at the surface in the simulations of 3 m, 2 m and 1 m water table depths were 0.019 mm/d, 0.031 mm/d and 0.118 mm/d, respectively.

In the previous simulations, the evaporation parameter h_A (i.e. the matric potential at which boundary conditions convert to Dirichlet pressure) was –40 m. This parameter was increased to –10 m and the preceding sandy clay simulation is re-run. The results indicate that the vadose salt transport predictions were relatively insensitive to this parameter, with surface salt concentrations of 0.001, 0.01 and 0.042 predicted after 189 days, 269 days and 365 days, respectively.

Water Table Salinities

The high salinity adopted in the previous simulations was indicative of either near-estuary or pumping-affected water table salinities. In order to gauge the possibility of vadose zone salinization for inland regions, two additional water table salinities (0.50 and 0.10) were simulated using a time-averaged water table depth of 1 m. Again, the hydraulic properties of sandy clay were assumed.

Simulated vadose zone salinities are illustrated for the 0.50-water table salinity case in Figure 4. As for the higher water table salinity, the initial stages of salt build-up were predicted at the ground surface. For the 0.50-water table salinity case, surface salt concentrations of 0.001, 0.01 and 0.024 were predicted after 203 days, 300 days and 365 days, respectively. In the 0.10 water table salinity simulation, surface salt concentrations of 0.001 and 0.0047 occurred after 263 days and 365 days, respectively. The linearity of the relationship between the surface salt concentrations and the water table salinities for these simulations was in agreement with the analytical expression for salt build-up under steady water movement by Elrick et al. (1994).

A Comment on Solute Dispersion

Dispersion was expected to have some influence on the prediction of surface salinization and vadose zone salt distributions due its influence on the thickness of the salt front (Elrick et al., 1994). Longitudinal dispersivities in soil columns have been estimated by various authors (eg. Grifoll and Cohen, 1996) and range from 0.045 m to 0.3 m. For unsaturated solute movements, longitudinal dispersion is expected to increase with decreasing soil moisture content (Grifoll and Cohen, 1996). The longitudinal dispersivity component of the effective dispersion adopted in HYDRUS-1D was saturation-independent, although the molecular diffusion component of the effective dispersion depended on tortuosity factors, which reduced with soil moisture content. Hence, solute dispersion in

HYDRUS-1D was essentially soil moisture-independent. This limitation was not explored further in this study. Given the other simplifications adopted in these simulations, it was not expected to significantly change the conclusions.

Two simulations were conducted using reduced dispersivity values (0.01 m and 0.000 1 m). Again, the soil was sandy clay, the time-averaged depth to the water table was 1 m and the forcing pressure wave magnitude was 1 m. The predicted 1-year relative salinities at the surface using dispersivities of 0.2 m, 0.01 m and 0.000 1 m were 0.045, 0.0015 and 0.000 68, respectively. Clearly, estimates of dispersion are important components of any study of the potential for surface salinization under evaporative stresses, as reductions in the dispersivity value virtually eliminated surface salt build-up in these simulations.

Soil Types
The water table wave magnitude varied for different soil types under identical boundary forcing head waves. Therefore, a comparison between different soil types using the same forcing wave characteristics would not isolate the impact of soil hydraulic properties on vadose zone transport. Hence, the forcing wave was modified in three different soil types: sand, sandy clay loam and sandy clay (Carsel and Parrish, 1988), until the water table wave in each simulation was similar (i.e. magnitude of 0.35 m and average depth of 1 m). In this way, the influence of soil type is isolated from water table wave magnitude effects.

In the simulations of sandy clay, sandy clay loam and sand, the forcing wave magnitudes were 1.329 m, 1.046 m, and 0.500m, respectively. Predicted vadose zone salt distributions are shown in Figures 5, 6 and 7. Clearly, the most susceptible soil to salt accumulation is the sandy clay loam (Figure 6), in which the 1-year surface salt concentration (relative) was 0.94.

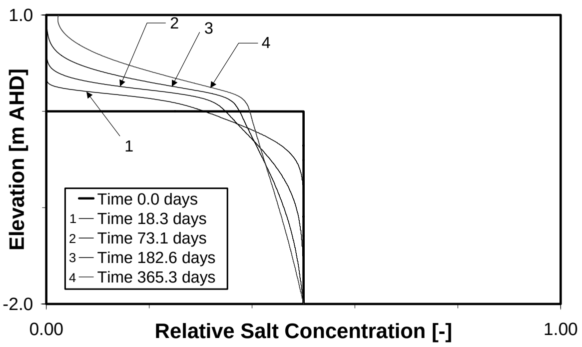

Figure 4. Vadose zone salinity changes for a water table salinity of 0.50.

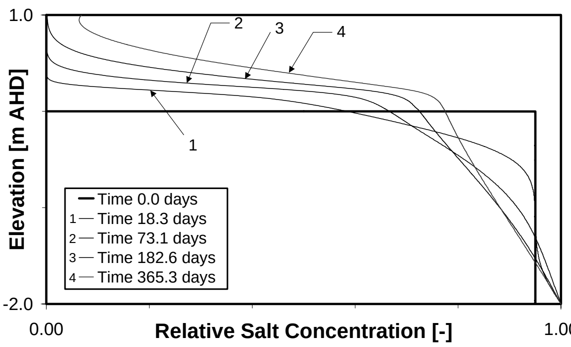

Figure 5. Vadose zone salinity changes in sandy clay for a water table wave magnitude of 0.35m.

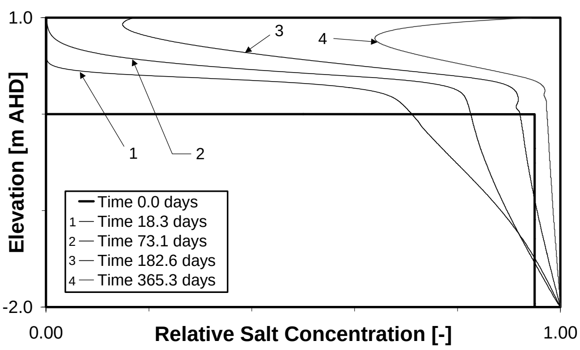

Figure 6. Vadose zone salinity changes in sandy clay loam for a water table wave magnitude of 0.35m.

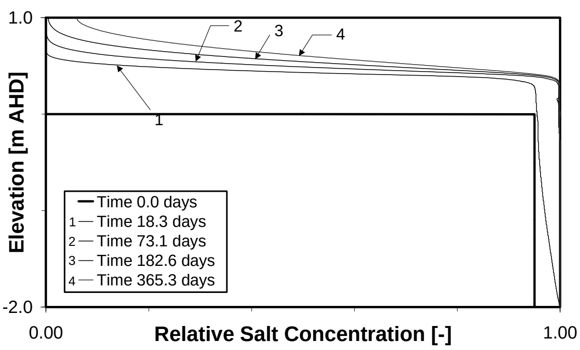

Figure 7. Vadose zone salinity changes in sand for a water table wave magnitude of 0.35m.

In the sand simulation, numerical oscillations, encountered with the original maximum time step length of 600 seconds were virtually eliminated by reducing the time step length to 40 seconds. The difference in prediction with the reduced time step was considerable in this case, highlighting the importance of analysing simulation results both temporally and spatially for such numerical errors. The reduced vadose zone salinization compared to the sandy clay loam case indicates that the potential for surface salt build-up is not merely related to the saturated hydraulic conductivity (highest in sand), but also to the moisture retention properties, particularly those contributing to the capillary rise.

Conclusions

In this paper, the potential for vadose zone and ground surface salinization from evaporation is explored for variations of soil type, water table depth, water table wave amplitude and water table salinity using numerical experiments of a tidally forced system using the code HYDRUS-1D. The near-estuary water table depth of 3m as encountered in some places in the Pioneer was clearly sufficient to avoid ground surface salt build-up during the simulation period of 1 year. Using the hydraulic properties for sandy clay, surface salts were observed building-up once the average water table depth was reduced to 1 m. The prediction of time-averaged water table depth and surface salinization is shown to be a function of the water table wave amplitude, as increasing the water table wave amplitude increased ground surface salinization under the same evaporation boundary conditions.

Simulations with reductions in the water table salinity indicated almost a linear relationship between the 1-year ground surface salinity and water table salinity. Thus, reducing water table salinity does not appear to completely halt the potential for salt build-up at the ground surface for the values considered (i.e. water table salinities of 0.1 to 1.0 relative to seawater); it merely reduces the rate of accumulation. The results presented in the current study are clearly dependent on the selection of solute longitudinal dispersivity. Ground surface salt build-up was almost eliminated by reducing the longitudinal dispersivity to 0.01 m and 0.000 1 m. Literature values of longitudinal dispersivity in similar systems range between 0.045m and 0.3m and as such, the value adopted here (0.2 m) is considered typical of field situations.

A comparison between different soil types was conducted by modifying the forcing pressure wave amplitude (and time-averaged pressure) until the water table wave amplitude was 0.35m and the water table depths were similar (1 m from the ground surface) in each case. In the three different soils, forcing wave amplitudes of 0.5 m (sand), 1.046 m (sandy clay loam) and 1.329 m (sandy clay) were required. Of the three soils, the sandy clay loam clearly showed the highest level of vadose zone and ground surface salinization. The sand and sandy clay soils produced similar ground surface salinization, with the sand soil simulation producing the lesser surface salinities of the two. Observation of predicted vadose zone salt distributions provides evidence that the higher saturated hydraulic conductivity of sand forces salts higher into the capillary zone than the other two soils, however the sharp decrease in the unsaturated hydraulic conductivity of sand limits the transportation of salts to the ground surface.

References

CSIRO Land and Water (2003). Salinity. Retrieved from CSIRO Land and Water website: http://www.clw.csiro.au/priorities/salinity/#how.
Gordon, I., B. Pearce, I. Heiner and A. Biggs (2002). Salinity hazard mapping in Qld. Department of Natural Resources, Mines and Energy Fact Sheet, Land Series, QNRM02270, Brisbane.
Pavelic, P., P.J. Dillon, K.A. Narayan, T.N. Herrmann, S.R. Barnett (1997). Integrated groundwater flow and agronomic modelling for management of dryland salinity of a coastal plain in southern Australia. *Agricultural Water Management* 35, 75-93.
Cook, P.G., F.W. Leaney and I.D. Jolly (2001). Groundwater recharge in the Mallee Region, and salinity implications for the Murray River – a review. CSIRO Technical Report No. 45/01, CSIRO Land and Water.
Werner, A.D. (2004). The interaction between a tidal estuary and a shallow unconfined aquifer: saltwater intrusion and environmental impacts in the riparian zone. Ph.D. thesis (under examination), University of Queensland, Brisbane.
Simunek, J., M. Sejna and M.T. van Genuchten (1998). The HYDRUS-1D software package for simulating the movement of water, heat, and multiple solutes in variably saturated media, version 2.0. U.S. Salinity Laboratory, ARS, USDA: Riverside, California, 178.
van Dam, J.C., and R.A. Feddes (2000). Numerical simulation of infiltration, evaporation and shallow groundwater levels with Richards equation. *Journal of Hydrology* 233, 72-85.
Carsel, R.F., and R.S. Parrish (1988). Developing joint probability distributions of soil water retention characteristics. *Water Resources Research* 24 (5), 755-769.
Murphy, S.F., and R.C. Sorensen (2000). Develop a water resource management strategy for the Mackay coastal aquifer system. Department of Natural Resources, Queensland Government, Mackay.
Cramer, V.A., and R.J. Hobbs (2002). Ecological consequences of altered hydrological regimes in fragmented ecosystems in southern Australia: Impacts and possible management responses. *Austral Ecology* 27 (5), 546-564.
Cook, F.J., and D.W. Rassam (2002). An analytical model for predicting water table dynamics during drainage and evaporation. *Journal of Hydrology* 263, 105-113.
Grifoll, J., and Y. Cohen (1996). Contaminant migration in the unsaturated zone: the effect of rainfall and evapotranspiration. *Journal of Contaminant Hydrology* 23, 185-211.
Zheng, C., and G.D. Bennett (2002). Applied contaminant transport modelling : theory and practice. Van Nostrand Reinhold, New York.
Department of Primary Industries (1994). Farm water supplies design manual: Volume 2 Irrigation Systems. Edited by A.J. Horton and G.A. Jobling. Rural Advisory Service, Department of Primary Industries, Brisbane.
Elrick, D.E., A. Mermoud, and T. Monnier (1994). An analysis of solute accumulation during steady-state evaporation in an initially contaminated soil. *Journal of Hydrology* 155 (1-2), 27-38.