5.2 Modelling Water and Salt Movement on the Chowilla Floodplain

W. R. Dawes, P. G. Slavich, T. J. Hatton and G. R. Walker

Abstract

River level management in the lower River Murray has had a profound negative impact on native floodplain vegetation. Reduced flooding, both amount and frequency, and rising levels of naturally saline groundwater have lead to salinisation of the soil and the subsequent reduction of tree density and health. To develop management guidelines there is a need to understand the effect of different flooding and watertable conditions on vegetation growth and salinisation processes.

Field studies of vegetation water use during and after a flood were used to calibrate a physically based soil–vegetation–atmosphere transfer model (WAVES). The model parameterisation was tested using long term simulations which also give insight into the development of dieback. Observed water content and chloride profiles have been reproduced after a flood event and after a drying cycle. Soil hydraulic properties were set initially using limited field measurement and soil textural descriptions. There is confidence in this approach, since calibrated parameters show remarkable consistency across five sites, with varying leaf area, groundwater depth, and time and depth of flooding. The calibrated model can be used to explore the effect of river management scenarios on existing riparian vegetation.

5.2.1 Introduction

In dry areas of Australia, flooding can be an important source of water for riparian vegetation. Where aridity is coupled with salinity, flooding can be a critical factor in supplying fresh water and leaching accumulated salts from the root zone. The native floodplain vegetation in the lower River Murray has declined in health as a result of river level management over the last few decades. A system of reservoirs and weirs (locally known as locks) was installed along the river during the 1920s to regulate the river flow for reliable year round water supply and navigation. The decline in the number of medium-sized floods has reduced salt leaching, and the installation of locks has resulted in raised saline watertables (Walker *et al.*, 1996). Guidelines for river and groundwater management to control soil salinisation at sites along the river are currently being developed. Margules and Partners *et al.* (1990) estimated that approximately 180 km^2 of floodplain along the River Murray is severely degraded, of which 53% has saline groundwater identified as the major cause of degradation.

The time scales for soil salinisation can be used to establish simple management guidelines and have been estimated using a steady state groundwater discharge model (Jolly *et. al.*, 1993). More complex physically based models which describe soil–vegetation–atmosphere transfers (SVAT models) can be used to increase understanding of the interactions between the vegetation growth and water use and processes affecting the movement of water and salt, and can explicitly take into account changed vegetation and river level management. Guidelines based on simple steady state models also need to be evaluated using more complex SVAT models.

This section describes the first stage in attempting to understand the interactions between vegetation and soil salinisation processes occurring on the Chowilla floodplain in the lower Murray River. Field data from a number of sites representing the range of floodplain conditions are used to calibrate the SVAT model WAVES (Dawes and Short, 1993). The hydrological responses of the sites to flooding is analysed in some detail, vegetation health is related to long-term flooding history, and is shown that it may be used to better predict the impacts of changed flood management.

5.2.2 Site Description

The study region is the Chowilla anabranch, a 200 km^2 area of semi-arid saline floodplain on the Lower River Murray, located on the South Australia–New South Wales border of Australia (Fig. 5.6). Jolly and Walker (1995) provide an excellent overview of the hydrology, hydrogeology, vegetation, and management problems of the area. Briefly, the area consists of a network of streams which flow from the River Murray upstream of Lock 6, across the floodplain before joining into Chowilla Creek, and discharging back into the River Murray downstream of Lock 6. Before the installation of the lock, these streams were ephemeral and flowed only in times of flood; they now carry up to 80% of the River Murray flow. This is an excellent study region for the following reasons: (1) there has been a significant body of data collected in the region during investigations for a proposed dam, and salt interception schemes, (2) it is the site of the second-largest natural salt load to the River Murray, approximately 50 000 t yr^{-1} (Walker *et al.* 1996), and (3) it is a wetland of international significance listed under the UNESCO Ramsar Convention (Section 14.5) (NEC 1988).

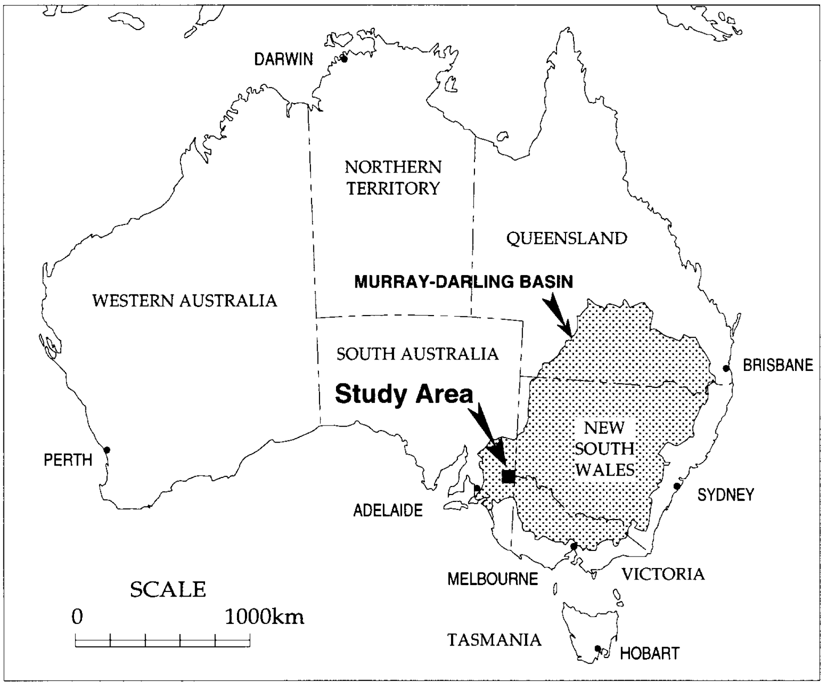

Fig. 5.6. Experimental site location map.

The Chowilla region has a semi-arid climate with annual average rainfall of approximately 250 mm yr^{-1}. This annual volume is highly variable, ranging from 100 to 500 mm yr^{-1}. Average annual potential evaporation is around 2000 mm yr^{-1} (Jolly *et al.* 1993). The composition and distribution of vegetation at Chowilla has been described by O'Malley (1990). The dominant species are the trees black box (*E. largiflorens* F. Muell.) and river red gum (*E. camaldulensis*), the shrub lignum (*Muehlenbeckia cunninghamii*), and large areas of annual grass. The distribution of these species is controlled by flood frequency, which is related to surface elevation, and groundwater salinity. Black box is located on higher areas of the floodplain, redgums occur generally along stream and creek courses, and lignum is found on clay fan features adjacent to streams. In this work we concentrate on black box, as it is the predominant species, and is known to be adversely affected by current river management (Margules and Partners *et al.*, 1990).

The soils of the Chowilla floodplain have been described by Hollingsworth *et al.* (1990). They consist generally of a layer of alluvial grey cracking clay, known as Coonambidgal Clay, up to 5 m deep, overlying an unconsolidated alluvial sand deposit, known as Monoman Sand, approximately 30 m deep. The boundary between these layers is often unclear, with transitional material of varying clay content up to 1 m thick. Groundwater levels have risen since the installation of Lock 6, from the Monoman Sand formation to between 2 and 4 m from the surface in the Coonambigdal Clay.

The hydrology of the area is described by Jolly and Walker (1995). Briefly, the frequency of medium-sized floods has decreased to about one third of natural conditions as a consequence of development of upstream storages. Black box trees are generally found in areas where the mean return period of floods is greater than 8 years. Enhancement of floods for environmental purposes is possible by the release of water from two nearby storages, Lake Victoria and Menindie Lakes.

5.2.3 Monitoring sites

From a survey of vegetation health on the Chowilla floodplain, large scale spatial patterns of health were found to correlate with flooding frequency, groundwater depth, and groundwater salinity (O'Malley, 1990; Taylor *et al.*, 1996). Vegetation health was poorest in areas with infrequent flooding, and shallow high salinity groundwater tables. Where groundwater was fresher, or more frequently flooded, or on higher ground, vegetation health was significantly better. The variables were entered as layers into a Geographic Information System (GIS) and five broad categories were identified. Five study sites, representing each of the GIS classes, were chosen for monitoring. Soil profiles were sampled in September 1993, before a flood which occurred in November and December, just after the flood in January 1994, and in April 1994 after a three month drying period (McEwan *et al.*, 1995). Soils were analysed for gravimetric water content (kg/kg) and water-soluble chloride. This sampling gave the widest possible range of moisture and soil-water salinities for model calibration. These sites are samples only, and may not be representative of the whole of the class, and each class should not be seen as homogeneous.

Four broad soil-texture groups based on clay content (Table 5.2) were identified on the floodplain and were used to characterise soil horizons both across and within the sites. The depth of each soil horizon was estimated from field sampling, and examination of the gravimetric water content profiles.

Table 5.2. Four soil textural classes observed as soil horizons at calibration sites.

Texture Code	Clay %
T1	< 15
T2	15 – 30
T3	31 – 45
T4	> 45

5.2.4 Recorded data

The monitoring period for data analysed in this paper was the six months from October 1993 to April 1994. Soil profiles were sampled from each site in October before a flood, in January immediately after the flood, and in April after a long dry period. Water content was determined gravimetrically, matric potential was determined by the filter paper method, and chloride content was determined by 1: 5 soil paste extracts.

Daily values of maximum and minimum temperature, average vapour pressure deficit, rainfall and total radiation were obtained from an automatic weather station at the site and one at Loxton. These data were used in WAVES to estimate evaporation and transpiration demand at each site. The watertable was logged for the duration of the monitoring period using capacitance probes at each site. Transpiration was measured using the heat pulse method (Swanson and Whitfield, 1981; Green and Clothier, 1988) for several weeks at a time over the larger monitoring period. The complete dataset description can be found in McEwan *et al.* (1995).

Four of the five sites, sites S1, S3, S4, and S6, had saline groundwater. The salinity of the groundwater used in WAVES was set to that measured in soil-water extracts from just above the watertable. Site S5, however, showed significant leaching of salt below the groundwater surface, and so fresh water, with salinity of 0.004 dS m^{-1} (equal to rainfall), was used at this site. This site is closer to a creek than the other sites, and we hypothesise that fresh creek water is flushing from below during and after the flood.

Table 5.3 Values and units of vegetation growth and response parameters for Black box.

Description	Value	Units
Canopy albedo	0.1	–
Soil albedo	0.15	–
Rainfall interception coefficient	0.001	m LAI^{-1} d^{-1}
Light interception coefficient	–0.42	–
Maximum carbon assimilation rate	0.01	kg C m^{-2} d^{-1}
Canopy conductance model slope value	0.7	–
Maximum plant available soil water potential	–350	m
IRM weighting of water relative to light	1.13	–
IRM weighting of nutrients relative to light	0.3	–
Temperature when growth rate is half optimum	10	°C
Temperature when growth rate is optimum	20	°C
Saturation light intensity	1200	μmoles m^{-2} d^{-1}
Specific leaf area	12	m^2 kg^{-1}
Salt sensitivity factor	1.0	–
Aerodynamic resistance	20	s m^{-1}

The vegetation parameters required by WAVES were set using a combination of literature (Hodges, 1992; Hatton and Dawes, 1993) and measured values (Table 5.3). The maximum plant-available soil-water potential was set from seasonal observations of predawn leaf-water potentials (Eldrige *et al.*, 1993). Predawn leaf-water potentials were measured using a pressure bomb and diurnal changes in stomatal conductance by porometry. WAVES does not calculate canopy aerodynamic resistance (r_a) dynamically, and the constant value in Table 5.3 corresponds to a wind speed of 1–2 m s^{-1} at 2 m above a canopy height of 6–12 m. The static leaf area index was estimated by Taylor (1993) as 0.23, 0.29, 0.28, 0.42, and 0.19 for sites S1, S3, S4, S5, and S6 respectively.

The flood depth and duration were measured using the depth of water in the river at Lock 6. The five monitored sites are at different elevations and were flooded for 17, 30, 78, 61, and 14 days respectively.

5.2.5 Model Calibration

All calibration was done manually, *i.e.* software that optimizes parameters for a least squares or other error criteria was not used. The calibration approach required a compromise between the degree a parameter could be adjusted for an individual site, and the degree of parameter variation across sites. Parsimony was also exercised when estimating the number of distinct soil layers within each soil profile.

For λ and C, a simulation was run and their values were adjusted until the modelled water content profiles showed the correct shape before and after the flood. This adjustment fitted the moisture retention curve, *i.e.* the ψ *vs* θ relationship, and helped to compensate for the properties of the surface clay. Fitting of K_s was done after λ and C were set. A simulation was run, and K_s adjusted, until the modelled salt fronts moved the observed distance after the flood.

The calibrated values of soil hydraulic parameters are given in Table 5.4. The calibrated values of λ and C were physically sensible, *i.e.* larger for the horizons with higher clay content. One of the more encouraging aspects of the work is the general consistency of the calibrated soil hydraulic parameters across the five sites, each with different leaf area, watertable level dynamics, and flood duration. We are therefore confident that the soil hydraulic properties are representative of the soils in the area of the observation sites.

There was good agreement in general between the measured and modelled profiles before and after the flood (Figs. 5.6 to 5.11 for sites S1, S3, S4, S5 and S6 respectively). However, the modelled salt in the 0–1 m layer in particular, was less than observed. This suggests that the leaching process in this layer is relatively inefficient and that not all the solute is readily mobile. This is not unexpected in aggregated clay soils. The assumption explicit in using Richards' equation that the soil matrix is rigid is inappropriate in the surface cracking clay, and will contribute to the enhanced modelled leaching.

Table 5.4. Calibrated values of BW parameters for each soil horizon at each site.

Depth m	Texture Code	K_s m day^{-1}	θ_s	θ_r	λ m	C
Site 1						
0.0–0.6	T3	0.006	0.36	0.1	0.5	1.05
0.6 +	T2	0.006	0.36	0.1	0.3	1.04
Site 3						
0.0–0.7	T4	0.002	0.4	0.1	1.0	1.10
0.7–1.0	T3	0.006	0.36	0.1	0.5	1.05
1.0–1.4	T2	0.006	0.36	0.1	0.3	1.01
1.4 +	T1	0.05	0.36	0.05	0.2	1.01
Site 4						
0.0–0.6	T3	0.002	0.36	0.1	0.5	1.04
0.6–0.9	T4	0.002	0.45	0.1	1.0	1.10
0.9 +	T1	0.05	0.36	0.05	0.2	1.01
Site 5						
0.0–0.6	T4	0.002	0.5	0.1	1.0	1.10
0.6–0.8	T2	0.003	0.4	0.1	0.5	1.05
0.8 +	T1	0.05	0.36	0.05	0.2	1.01
Site 6						
0.0–0.3	T3	0.002	0.45	0.1	0.6	1.10
0.3–0.75	T4	0.002	0.45	0.1	1.0	1.10
0.75–1.8	T3	0.002	0.45	0.1	0.5	1.10
1.8 +	T4	0.002	0.45	0.1	1.0	1.10

The plant growth model had two fitted parameters. The slope parameter g_1 was calibrated to give good results on measured transpiration rates and soil-water profiles. The leaf carbon partitioning factor was calibrated to give a realistic range of leaf mass consistent with that observed for the 25 year simulations. Fig. 5.12 shows measured and modelled rates of transpiration at Site 1, from October 1993 to March 1994. While we expect the magnitude of transpiration to be right, the figure shows the model feedbacks reproduces well the changes in transpiration rates due to the flood event, and changing season.

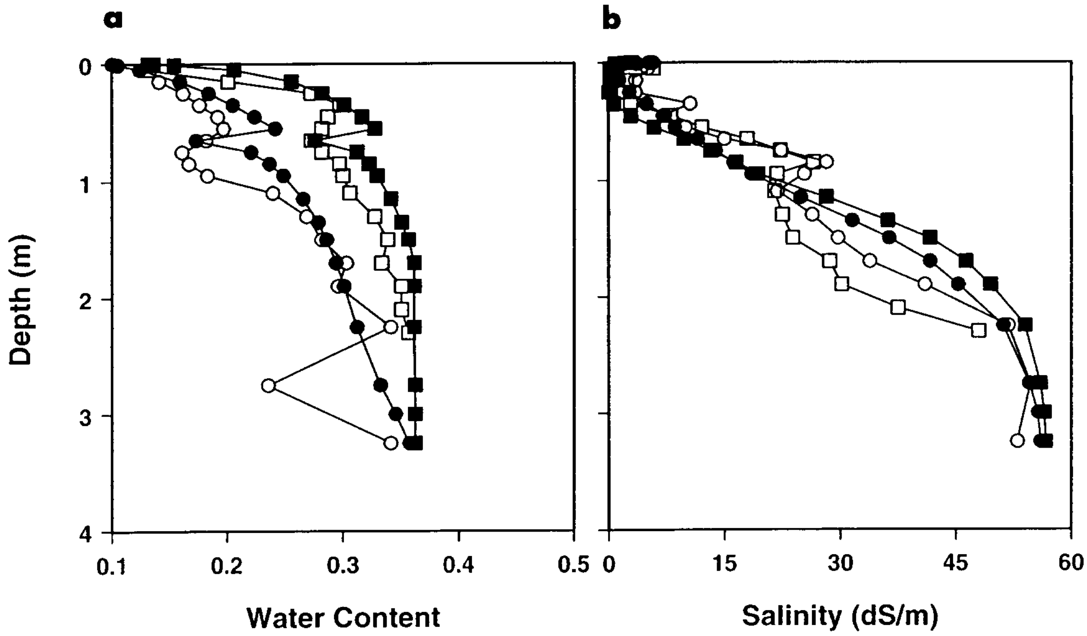

Fig. 5.7. Observed (open symbol) and predicted (filled symbol), (a) water and (b) salt profiles, from Site 1, for pre-flood (circles, January 1994) and post-flood (squares, April 1994) conditions.

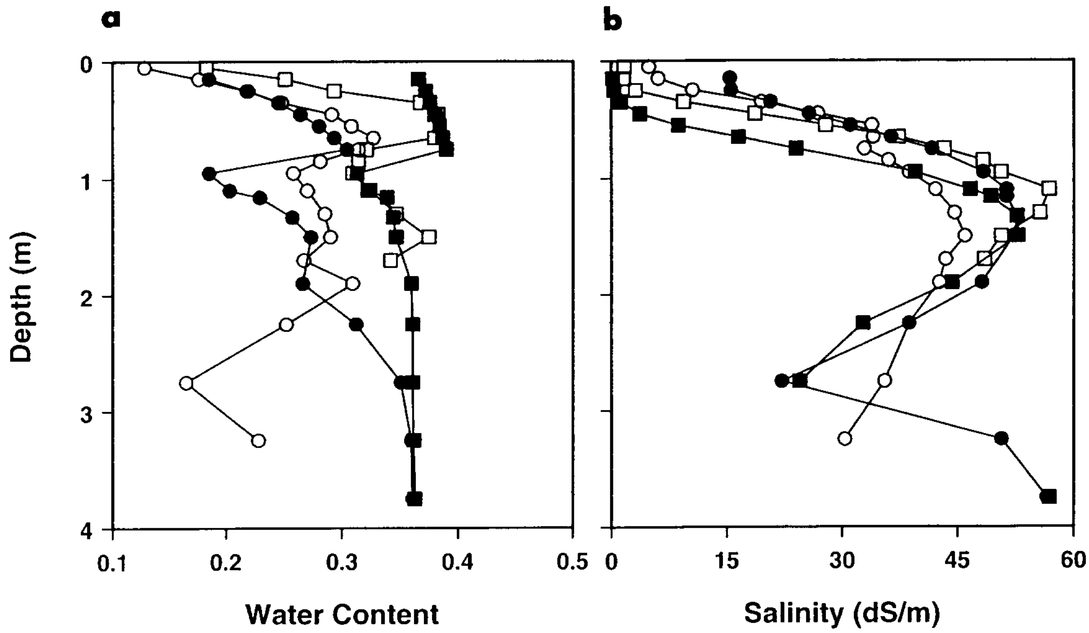

Fig. 5.8. Observed and predicted, (a) water and (b) salt profiles, from Site 3, for pre-flood (January 1994) and post-flood (April 1994) conditions.

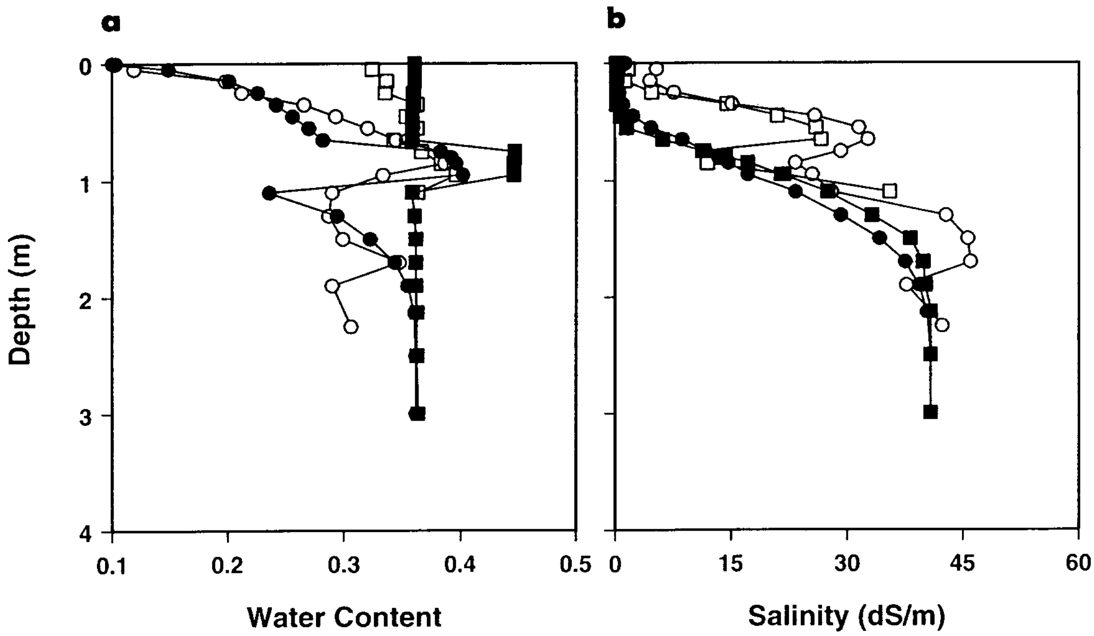

Fig. 5.9. Observed and predicted, (a) water and (b) salt profiles, from Site 4, for pre-flood (January 1994) and post-flood (April 1994) conditions.

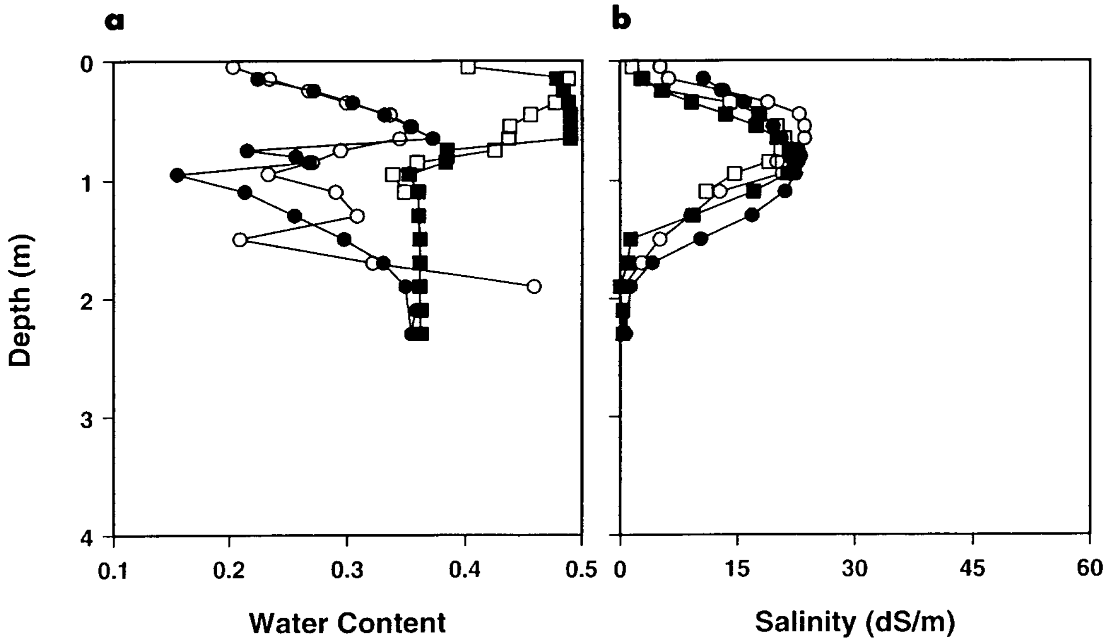

Fig. 5.10. Observed and predicted, (a) water and (b) salt profiles, from Site 5, for pre-flood (January 1994) and post-flood (April 1994) conditions.

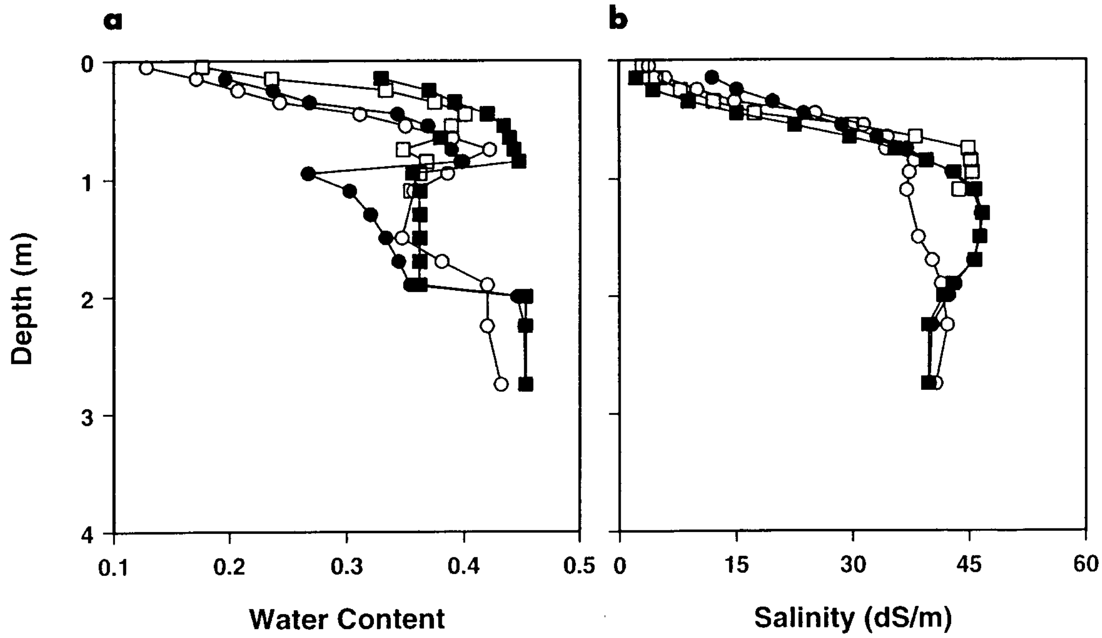

Fig. 5.11. Observed and predicted, (a) water and (b) salt profiles, from Site 6, for pre-flood (January 1994) and post-flood (April 1994) conditions.

As a further test of the calibration parameters, WAVES was run with observed groundwater and climatic data until January 1995 when the sites were monitored again. Fig. 5.13 shows the observed water and salt profiles at Site 1, and the model results using the original two soil layer description and a four soil layer model. It is apparent there is a lens of material at 0.3 m with different water holding properties to the two original layers; the water content changes but the measured matrix potential does not, indicating the profile is uniformly at residual water content. There is no better agreement with salt profiles using the four layer model because the movement of salt has been halted by the absence of water to move it.

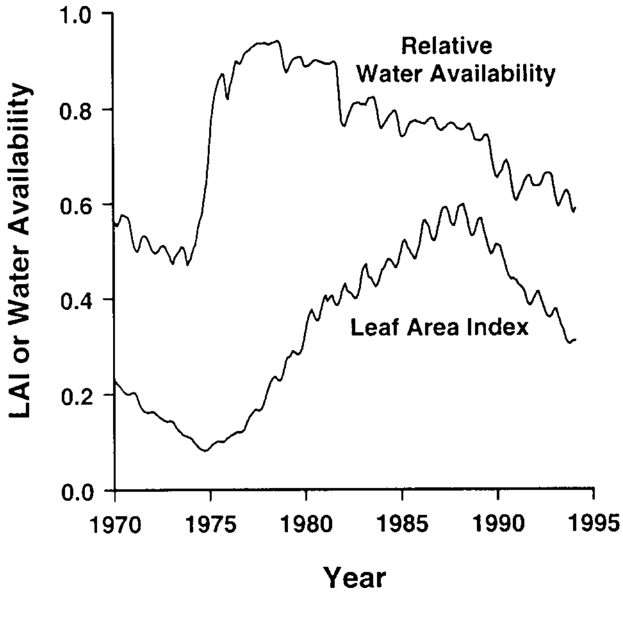

Fig. 5.12. Measured and modelled transpiration at Site 1 before and after the flood event.

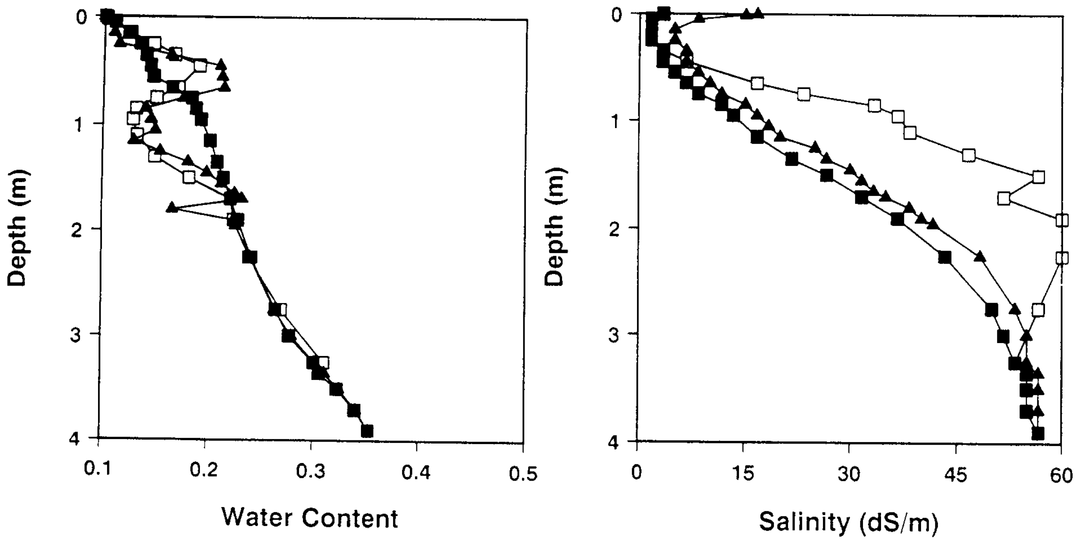

Fig. 5.13. Observed (open squares) and predicted, (a) water and (b) salt profiles, at Site 1 using two (filled squares) and four (filled triangles) soil layers for January 1995.

This does raise an important philosophical question for the modeller: what detail is required? If we are interested in short term flood dynamics to examine flood leaching events and closely matching profiles of salt and water, for example, then four layers would be appropriate. If we are interested in long term dynamics, retaining the important feedbacks and vertical detail, on longer term effects of salinisation on vegetation health, then use of two layers is adequate. We must also confront the question of the destructive nature of the sampling for salt and water profiles. The observed data approximate because of heterogeneity, so that exact matching is problematic.

5.2.6 Long Model Runs

The soil hydraulic and vegetation parameters were calibrated using a relatively short period of 180 days. The model parameters were further tested using a 25 year simulation run, from 1970 to 1995, to compare the response of the system to measured climatic and river dynamics at two sites. The long term behaviour should be consistent with field observations of vegetation decline in relation to salinity, and numerically stable. These simulations also illustrate the potential applications of the calibrated model. Sites 1 and 6 were used in the simulations, because they show strong contrast in soil type while having the same elevation and flooding extent. The main features to observe in the simulations are: (1) the development of a moving salt front from the deep watertable, which is consistent with field observation, (2) a 5 to 10 year period of vegetative decline from a shallow saline watertable; this is again consistent with field observations at shallow watertable sites, and (3) relatively long periods of consistent leaf area that drop within two years to a new level; this behaviour is consistent with Hatton and Wu (1995) who suggested that trees retain leaf area until sufficient stress has built up to necessitate a dramatic loss of leaves.

Fig. 5.14a shows the simulated leaf area and water availability scalar (used in the plant growth model) over 25 years at Site 1. Floods inundated the sites for a total of 219 days between 1974 and 1977. They are clearly shown in the water availability scalar, when much of the salt in the root zone is flushed down to the groundwater table. The water availability remains relatively high, around 80%, until 1990; this corresponds with a period of slow leaf area development. However, as salt builds up in the root zone (Fig. 5.14b), and especially in the top 2 m of soil, both water availability and leaf area show alarming decline. The only other flood of note is that in 1994 studied in this work. This site is currently rated as relatively healthy (Walker *et al.*, 1996, Ch. 10).

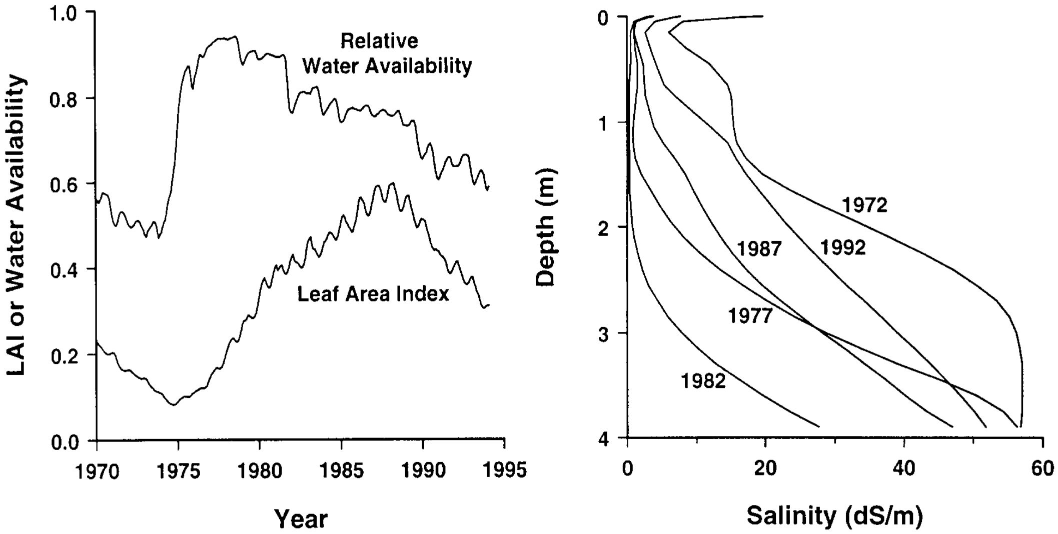

Fig. 5.14. Results of a 25 year simulation at Site 1, showing (a) water availability scalar and leaf area index, and (b) profile of soil water salinity.

Fig. 5.15a shows the simulated leaf area and water availability scalar over 25 years at Site 6. The soils at this site are heavier than Site 1, and the groundwater salinity is less. The net result is that the flood has resulted in only limited leaching of salt and only to 2 m depth. However, the consequent salt build up is mainly in the surface soil where a large bulge has developed. The leaf area and water availability graphs show little variation over the 25 years of simulation, indicating that the heavy soil acts as a buffer to both flooding and salt accumulation. Fig. 5.15b shows a salinity bulge forming in the profile between 1 and 2 m depth. This is consistent with the field observations made for the calibration exercise, and corresponds to a natural soil layer of low sorptivity at this site. Again note the steady build-up of salinity over time.

5.2.7 Comparison with other Models and Data

The hydraulic parameters shown in Table 5.4 were used to calculate maximum steady-state groundwater discharge with a watertable at 4.0m, after the method of Jolly *et al.* (1993). From Site 1, rates vary from 6.1 to 7.0 mm yr^{-1}, which compare favourably to Jolly's estimated value of

8.0 mm yr^{-1} from deuterium profiles from a bare site, and our modelled rates of 3.6 mm yr^{-1} for the vegetated site. This result is very encouraging since it was not calibrated directly: it is a result of calibrating transpiration and modelling soil-water dynamics realistically.

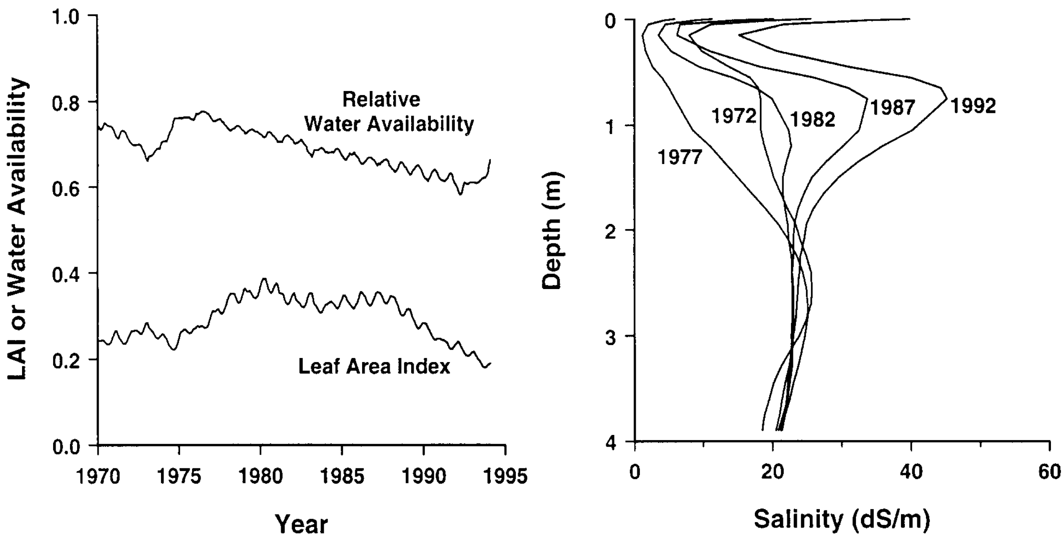

Fig. 5.15. Results of a 25 year simulation at Site 6, showing (a) water availability scalar and leaf area index, and (b) profile of soil water salinity.

The WAVES growth model requires a parameter that is the maximum proportion of gross assimilate partitioned to above-ground carbon pools, *i.e.* leaves and stems. The final actual amount is a function of this value and the water availability; less available water causes more resources to be devoted to root development. The calibrated maximum partitioning of 17% to leaves, 17% to stems, and 66% to roots is remarkably consistent with McMurtrie's (1985) partitioning of 20:20:60 for a "poor quality wooded site". Such a result shows that the internal feedbacks within WAVES reasonably represent the processes. The calibrated slope of the modified Ball *et al.* (1987) equation yielded canopy resistance of 120–4500 s m^{-1}, and averaging 330 s m^{-1}. McNaughton and Jarvis (1991) reported resistances of 50 s m^{-1} for well watered crops and pastures, around 100 s m^{-1} for forest and wild vegetation, and resistances of 250 s m^{-1} or more "in arid lands where leaf area index is very small, or the vegetation is suffering severe water stress". These reported values compare very well with those modelled with WAVES.

5.2.8 Conclusions

The parameterisation of the physically based ecohydrological model WAVES, has given good prediction of the rate of soil drying after flooding, the rate of chloride leaching during flooding, and the rate of transpiration for a 6-month monitoring period. The good performance of WAVES in fitting both water and salt profiles is encouraging. If this exercise were performed for a single site only, then these results would have less significance. However, the good performance at a range of sites with different soil layering, flooding and vegetation covers is significant for this type of model.

Empirical models of salt and water balances generally do not give vertical distributions of water and salt at a site, and do not easily handle changing groundwater levels or floods. Further, their fitted parameters may vary greatly between sites such as these. However, if point-based SVAT models can identify critical parameters and processes of interest, in the future simpler models may be developed that only model these critical feedbacks but at the whole floodplain scale, say within a GIS framework, and may be compared to remotely sensed data (Taylor *et al.*, 1996).

The utility of the calibrated model is clear. It can be used to explore the likely impacts of flooding patterns, along with the time scale of plant response. This can aid in the definition of a critical groundwater depth, or flooding frequency, for a given level of plant health or cover. All of these results may be site dependant also, so that the mean return period required to keep vegetation at Site 6 "relatively healthy", with water availability at 30% or greater for example, may be greater than for Site 1, which might be shown to have more rapid salinisation. The experience at Site 5 with a fresh water lower boundary condition, has also given useful insight into the processes occurring near creeks on the floodplain.

This paper is the first in a series from continuing work at Chowilla and other floodplain areas along the River Murray. Othe papers describe in detail (1) development of the new plant growth model (Slavich *et al.*, 1998), (2) alternate management scenario modelling using the calibrations established here, and (3) comparison of WAVES with a simpler lumped model of water and solute behaviour to evaluate critical processes, and how well they can be reproduced by simpler means.

Engineering for human benefits has had a dramatic negative effect on health of riparian vegetation. The decisions on river management are being made now by committees responsible for areas along the river course, such as the Chowilla Working Group. The ability to evaluate and plan management regimes for the best ecological benefits, may ultimately result in the stabilisation and conservation of these fragile and important areas.

Acknowledgements

The authors wish to thank Ian Jolly, Lisa Mensforth, Peter Taylor, and Peter Thorburn for their efforts collecting data at Chowilla for this, and other, papers.