



Groundwater - surface water interactions in arid/semi-arid wetlands and the consequences of salinity for wetland ecology

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Review prepared as a component of the South Australian Centre for Natural Resources Management Project 054121: Surface Water - Groundwater Interactions in River Murray Wetlands and Implications for Water Quality and Ecology



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Executive Summary

Background

Wetland ecology depends on water regime and water quality. In arid/semi-arid environments average annual rainfall is seasonal, highly variable and significantly less than evaporation. Groundwater discharge can be a major component of the water balance in these environments. It can also be a significant source of salt and hence a major determinant of wetland ecology. Under natural conditions, wetlands in arid/semi-arid zones may periodically have experienced increases in salinity as a consequence of the high evaporative conditions and the variability of water flow. However, due to the impacts of human population pressure and the associated changes in land use, surface water regulation and water resource depletion (recognised as the fundamental causes of wetland loss and degradation worldwide), wetlands in arid/semi-arid environments are experiencing extended periods of salinity.

The role of wetlands in the hydrological cycle has been well documented (see the comprehensive review of Bullock and Acreman (2003)) and increasingly, groundwater and surface water are being treated as one system in hydrological studies (Winter *et al.*, 1999; Winter, 1999; Winter, 2001; Hayashi and Rosenberry, 2002; Sophocleous, 2002). However, arid/semi-arid wetlands remain poorly understood particularly in relation to the contributions that groundwater makes to the salt and water balance of the wetland. Recent literature has identified the importance of multi-disciplinary studies, particularly ecohydrology (see Hannah *et al.*, 2004). However, ecohydrology has not traditionally focused on the role that groundwater plays in wetland environments. Whilst this is changing (see for example the papers in Baird *et al.*, 2004) there is still very little literature relating to saline groundwater environments and their impacts on wetlands.

This report reviews the current knowledge of the role that surface water-groundwater interactions play in semi-arid wetlands with emphasis on:

- surface water / groundwater interaction;
- water and salt balances of wetlands; and
- ecological responses to changes in salinity as a consequence of the changes in both surface water and groundwater regimes.

Conclusions

The few available studies suggest that in arid/semi-arid areas groundwater/surface water interactions in wetlands are highly dynamic, are both temporally and spatially complex, and often extend beyond the surface water boundaries of the wetland. In areas where groundwater is low in salinity, it has beneficial impacts on wetland ecology which can be diminished in dry times when groundwater levels and hence inflows to wetlands are reduced or even cease. Conversely, if groundwater is saline, and inflows increase due to raised groundwater levels caused by factors such as land use change and river regulation, then this may have detrimental impacts on the ecology of a wetland and its surrounding areas.

Groundwater/surface water interactions in wetlands are mostly controlled by factors such as differences in head between the wetland surface water and groundwater (which in turn are controlled by climate, catchment land use and river management), the local geomorphology of the wetland (in particular the texture and chemistry of the wetland bed and banks), and the wetland and groundwater flow geometry.

Groundwater/surface water interactions in wetlands can be broadly classified into three flow regime types: (i) recharge - wetland loses surface water to the underlying aquifer; (ii) discharge - wetland gains water from the underlying aquifer; or (iii) flow through - wetland gains water from the groundwater in some locations and loses it in others. However, it is important to note that individual wetlands may temporally change from one type to another

depending on how the surface water levels in the wetland and the underlying groundwater levels change over time in response to climate, and catchment and river management.

The salinity in wetlands of arid/semi-arid environments will vary naturally due to high evaporative conditions, sporadic rainfall, groundwater inflows, and freshening after rains or floods. However, wetlands are often at particular risk of secondary salinity because their generally low position in the landscape exposes them to increased saline groundwater inflows caused by rising water tables. Terminal wetlands are potentially at higher risk than flow through systems as there is no salt removal mechanism.

There has been almost no modelling of groundwater/surface water interactions in arid/semi-arid wetlands with respect to water fluxes, let alone salinity or ecology. There is a clear need to develop modelling capabilities for the movement of salt to, from and within wetlands so to provide temporal predictions of wetland salinity which can be used assess ecosystem outcomes.

There has been a concerted effort in Australia to collect and collate data on the salinity tolerance/sensitivity of freshwater aquatic biota and riparian vegetation. There are many shortcomings and knowledge gaps in these data, a fact recognised by many of the authors of this work. Particularly notable is that there is very little time series data. This is a serious issue given that wetland salinities are often highly temporally variable due to changes in surface water inflows and outflows, evapoconcentration, and fluctuations in groundwater inputs and losses. There is also a concern that many of the data are from highly controlled laboratory experiments which may not represent the highly variable and unpredictable conditions experienced in the field. In light of these, and many other shortcomings identified, our view is that the data are a useful guide but must be utilised with some caution.

Secondary salinity can impact on wetland biota through changes in both salinity and water regime which result from the hydrological changes associated with secondary salinity. Whilst there have been some detailed studies of these interactions for some Australian riparian tree species, the combined effects on aquatic biodiversity have not yet been fully elucidated, and therefore a future research need.

Rainfall/flow pulses are a strong control on ecological function in arid/semi-arid areas. They also have an important indirect role through their impacts on wetland salinity. Freshwater pulses can be the primary means by which salt stored in both the water column and in the underlying sediments are flushed from wetlands. Conversely, increased runoff is also a commonly observed consequence of secondary salinity and so wetlands can experience increased surface water inflows which are often higher in salinity than under natural conditions. Moreover, changes in rainfall/flow pulse regimes can have significant impacts on wetland groundwater/surface water interactions. It is possible that in some instances that the groundwater inflow to a wetland may become so large that it could become a major component of the water balance and hence mask the role of natural pulsing regimes. However, if the groundwater is low in salinity it may provide an ecological benefit in arid/semi-arid areas by assisting in maintaining water in wetlands that become aquatic refugia between flow pulses.

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1. Introduction

1.1. Background

Estimates of the extent of wetlands systems globally are uncertain, due in part to confusion over what constitutes a wetland and the difficulties of delineating and mapping habitats with variable boundaries (Finlayson and D'Cruz, 2005). This is particularly pertinent in arid/semi-arid areas where the majority of wetlands are temporary. What is certain though is that a large percentage of wetlands have been lost in the last century and that ongoing degradation and loss is occurring worldwide (Williams, 1999). Whilst there are many causes, including drainage and land clearance, they are all related to agricultural, urban and industrial development associated with human population pressure (Finlayson and D'Cruz, 2005).

Subsequently, there has been increasing research into the function and value of wetlands. The role of wetlands in the hydrological cycle, for flood mitigation, biogeochemical functions, as nutrient and pollution filters for water quality improvement, and maintenance of food webs as habitats for a diverse range of biota has been well documented. Indeed wetlands are increasingly being utilised to enhance these values, including the re-creation and development of new, "artificial", wetlands. Over the past 15 years there has also been interest in the economic value of wetlands for the provision of these services along with fisheries production and wetland ecology, and its value for tourism. At the same time there is increasing demands being made on water supplies, and concerns over climate change and the expected increase in extent and distribution of arid/semi-arid areas.

As highlighted by a recent review by Sophocleous (2002) there has been increasing attention given to the interactions between groundwater and surface water (GW-SW). Similarly, the review of Danielopol *et al.* (2003) highlights the linkages between groundwater and ecosystems and the ever-increasing human-induced pressures on groundwater systems that have flow on ecological impacts. In the case of wetlands the majority of the GW-SW research related to wetland ecology appears to have been in temperate and tropical environments. Wetlands in arid/semi-arid areas have all the problems of temperate and tropical areas, i.e. pollution, drainage, eutrophication and changes to hydrological regime, including surface water impoundment and diversion and groundwater extraction. But they are also prone to salinisation due to human induced changes to the hydrological cycle. Available surface water is declining and the over-pumping of groundwater beyond natural recharge rates is occurring, lowering the water table and causing an increase in groundwater salinity and ecological degradation. The African and Australian continents have the largest arid/semi-arid areas and the largest salinised areas in the world (Ghassemi *et al.*, 1995).

The role of wetlands in the hydrology of arid/semi-arid environments is still poorly understood, particularly GW-SW interaction. In arid/semi-arid areas, where rainfall is variable, wetland ecosystems provide vital habitat for unique biota in an otherwise dry environment. Due to the temporary nature of many wetlands, and the resultant variability of physico-chemical factors, the biota have evolved unique features and life cycle adaptations that enable them to persist over dry intervals. Arid/semi-arid wetlands show faunal diversity as high as or greater than in many temperate and tropical wetlands. This is due to the evolutionary path (sea → rivers → river pools → temporary fresh wetlands) in which the biota in arid zones have evolved (Williams, 1998a; Williams, 1998b).

Paradoxically, the arid/semi-arid areas contain some of the world's largest river systems (for example, the Colorado, Nile, and Murray-Darling River systems) as the source of the rivers are from wetter areas (Williams, 1998a). Also, some of the most important wetlands of the world are in the arid/semi-arid zone including the Okovango Delta (Botswana), the Kafue Flats (Zambia), the Hadejia-Jamaare (Nigeria) and the Prairie Potholes (North America) (Kingsford, 1997). However, this is not reflected in the degree of knowledge of arid zone wetlands or in their conservation status (Kingsford, 1997).

Developments in hydrological research have seen surface water and groundwater increasingly being treated as part of the same system (Winter *et al.*, 1999; Winter, 1999;

Winter, 2001; Hayashi and Rosenberry, 2002; Sophocleous, 2002) and it is increasingly recognised that sustainable management of wetlands systems requires a multidisciplinary approach which includes both GW-SW hydrology and ecology (ecohydrology or hydroecology). Traditionally ecologists have not looked far below the surface, and hydrology has often been studied in isolation from ecology. Most studies of the impact of salinity on water bodies have concentrated on water supply impacts and only recently has the impacts on ecology been considered.

This review aims to bring together the state of knowledge of hydrological processes, GW-SW interaction, and the role of groundwater in wetlands of arid/semi-arid areas. Whilst this is a review of research literature worldwide, most of the literature on the ecological impacts of salinity is from Australia where research in this area is more advanced due to the extent of salinisation in that country. However, it is expected that many of the concepts developed from the Australian experience will apply to arid/semi-arid wetlands in other areas of the world that are influenced by groundwater.

1.2. Scope of Review

1.2.1. Climate

This review is concerned only with wetlands in arid/semi-arid environments. These regions receive low ($< 500 \text{ mm yr}^{-1}$ in winter rainfall areas and $< 800 \text{ mm yr}^{-1}$ in summer rainfall areas) and variable annual rainfall and are characterised by potential evapotranspiration that is far greater than precipitation (ratio of mean annual precipitation to mean potential evapotranspiration < 0.5 ; United Nations Environment Programme, 1992). For example, in Southern Australia annual evapotranspiration exceeds rainfall by more than 1500 mm. Extreme climatic variability and subsequent hydrological fluctuations are typical in these regions. The climatic variability occurs seasonally, interannually and on longer time frames. Consequently the arid/semi-arid areas are subjected to frequent and severe droughts and infrequent but significant floods. The climate variability and subsequent fluctuating hydrology are the key drivers of ecology in arid/semi-arid environments. (Many forms of wetlands are excluded from this review on the basis of climate for example, fens, peatlands, mires, and bogs, all of which occur in temperate and tropical climatic zones).

1.2.2. Wetland Types

Wetlands are generally described by the surface water regime. Hence there are many descriptors for wetlands, ranging from permanent through to variations of temporary, including seasonal, intermittent, episodic and ephemeral. Subsequently, the variety of wetlands in arid/semi-arid areas is greater than other climatic regions as the periods of inundation fluctuate between permanent and various temporary states.

The wetlands of interest in this review are naturally occurring inland shallow ($< 5 \text{ m}$) standing (lentic) freshwater wetlands of arid/semi-arid environments covering the spectrum of inundation from permanent to various temporary states. Herein we refer to wetlands as permanent and temporary as per Williams (1998a). Permanent wetlands contain water all year except in extended drought. Temporary includes intermittent wetlands, which have a wet or dry annual cycle, and episodic, which fill unpredictably.

This review is confined to naturally occurring (not constructed) freshwater wetlands that are subject to salinisation induced by human activity (secondary salinity) as opposed to naturally occurring primary salinity which occurs in endorheic systems (closed basins). Whilst it is recognised that chemical elements other than salinity may be toxic in wetlands, for instance pH, dissolved oxygen, or the concentration of specific ions (for example Na^+ , K^+ , Mg^{2+} , Ca^{2+} , SO_4^{2-} , HCO_3^- , CO_3^{2-} etc.), salinity is the focus of this review. Of interest here is the increases in salinity which change naturally freshwater wetlands (Total Dissolved Salts, TDS, of less than $1,000 \text{ mg L}^{-1}$, as per Davis *et al.* (1993)) into saline wetlands.

It is also recognised that there are other environmental factors that impact on ecology including temperature (thermal variation resulting from different sources of water, for instance groundwater has a relatively constant temperature, high water temperatures in summer as

water evaporates results in low concentrations of oxygen), nutrients (hillslope groundwater seepage is nutrient poor compared with river-derived groundwater seepage which is nutrient rich (and fresher), and turbidity (groundwater characterised by low suspended solids (Hayashi and Rosenberry, 2002; Amoros and Bornette, 2002). While all may be affected by groundwater, the focus of this review is on two aspects, water regime (surface water and groundwater) and water quality (specifically salinity).

2. Groundwater – Surface Water Interactions and Implications for Wetland Salinity

In this chapter we review the current understanding of the processes of GW-SW interaction which occur in arid/semi-arid wetlands and the implications for the salinisation of wetlands. We commence by defining some of the key relevant groundwater processes such as flow systems, recharge and discharge. Next we discuss some of the controls on GW-SW interaction in wetlands. We then discuss the processes of wetland salinisation. This is then followed by a discussion of relevant studies of the water and salt balance in wetlands. Finally, modelling approaches used for describing GW-SW interactions in wetlands are discussed.

2.1. Key Groundwater Processes

Hayashi and Rosenberry (2002) provide an excellent description of the fundamental concepts of GW-SW interaction and the implications for ecology. Some of the information they present is summarised here in the context of arid/semi-arid wetlands. We first define some key groundwater terms (a complete list is given in the Glossary):

Aquifer: a saturated permeable soil or geologic strata that can transmit significant quantities of groundwater under a hydraulic gradient.

Discharge: the loss of water from an aquifer (i) to the atmosphere by evaporation, springs and/or transpiration, or (ii) to a surface water body (in the case of rivers it is generally referred to as base flow) or the ocean, or (iii) by extraction.

Groundwater: sub-surface water in soils and geologic strata that have all of their pore space filled with water (i.e. are saturated).

Hyporheic zone: the saturated zone of mixing between groundwater and surface water beneath surface water bodies.

Recharge: the addition of water to an aquifer, most commonly through infiltration of a portion of rainfall, surface water or irrigation water that moves down beyond the plant root zone to an aquifer.

Groundwater systems are dynamic three-dimensional flow fields where movement of groundwater is driven by potential gradients (usually described by hydraulic heads) from areas where water is added to the aquifer (recharge) to areas where it is lost from the aquifer (discharge). The flow fields can be comprised of different sizes and depths and can overlie one another (Figure 1). Local flow systems are the most dynamic and the shallowest and therefore have the greatest interaction with surface water bodies. An example of a local flow system is shown in Figure 2 where a typical wetland is incised within a floodplain riparian zone which is the transition between the wetlands (and rivers) and surrounding upland areas (Figure 2).

Because the shape of the water table often replicates the shape of the land surface, it is generally shallow beneath the riparian zone and deeper below the upland areas. However, groundwater flow directions are governed by the slope of the water table, and so if the elevation of the water table under the upland area is higher than that beneath the riparian zone then groundwater flows into the riparian zone. Conversely, if the elevation of the water table beneath the riparian zone is higher than that below the upland area the groundwater will flow from the riparian zone into the upland area.

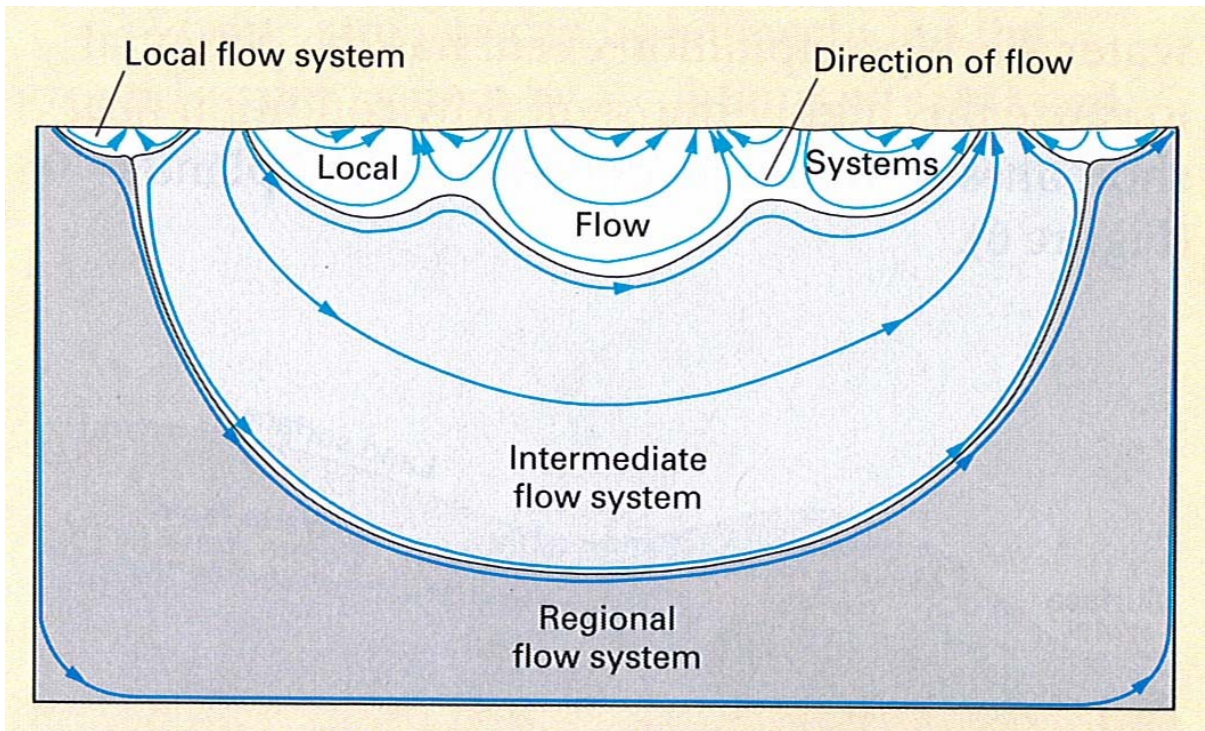


Figure 1. Groundwater flow systems can occur at local, intermediate, and regional scales. Most discharge of groundwater into wetlands is from local flow systems (Figure from Winter et al., (1999) who modified a similar figure from Toth (1963).

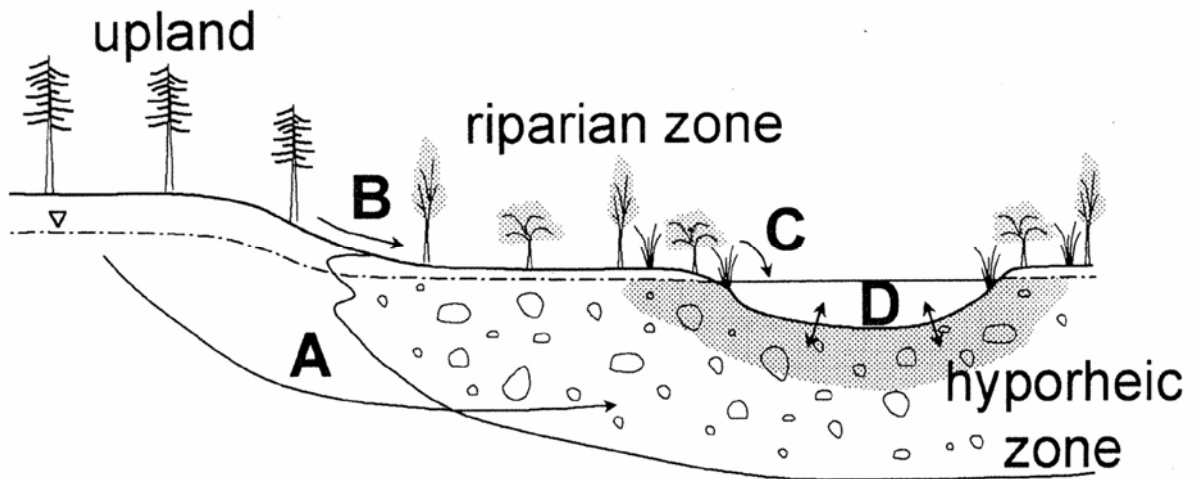


Figure 2. Schematic cross-section of a wetland, hyporheic zone (shaded), riparian zone and upland. Major pathways of water and constituent exchange are indicated by (A) groundwater flow, (B) overland flow, (C) litter fall, and (D) hyporheic exchange (Figure from Hayashi and Rosenberry (2002).

The flow of groundwater between the riparian zone and the wetland (via the hyporheic zone) will depend on the relative elevations of the water table beneath the riparian zone and the surface water in the wetland. Amoros and Bornette (2002) point out that GW-SW interactions can be very dynamic in the short-term as a result of varying river and wetland water levels (Figure 3). In the long-term groundwater exchange directly affects the ecology of surface water by sustaining stream base flow and moderating water level fluctuations of groundwater fed water bodies such as lakes and wetlands (Hayashi and Rosenberry, 2002). This is particularly the case in arid/semi-arid environments, where surface water regimes are vulnerable to rainfall variability and/or river regulation and abstraction activities, and so the persistence of wetlands can be dependant either completely or partially on contributions from groundwater. Groundwater can therefore be a major component of water balance of wetlands in arid/semi-arid areas. Indeed, wetlands can be completely groundwater dependant, with no surface expressions of water (for example the mound springs of the Great Artesian Basin in central Australia; Mudd, 2000). Many ecosystems survive solely on groundwater (Hatton and Evans, 1997). Conversely, if groundwater is saline, as it commonly is in arid/semi-arid areas, then it may have a detrimental impact on wetland ecology.

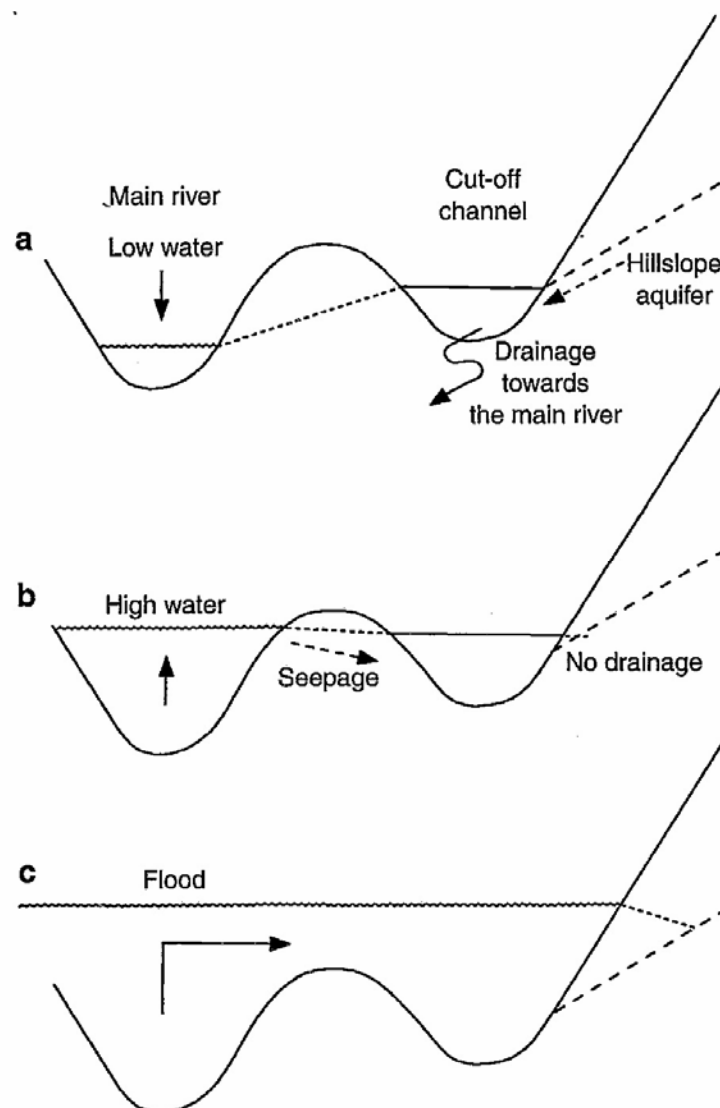


Figure 3. Schematic illustration of the short-term dynamics of GW-SW connectivity in relation to river stages: (a) (low-water stage), the wetlands may be supplied by a hillslope aquifer; (b) (high-water stage), the wetlands are supplied by river infiltration into the alluvial aquifer and possibly by river backflow through a downstream connection; (c) (flood), the wetlands are supplied by overbank flow (Figure from Amoros and Bornette, 2002).

2.2. Controls on Groundwater – Surface Water Interactions in Wetlands

As described above, GW-SW interactions of wetlands are controlled by the relative surface water and groundwater heads which can vary significantly over the short-term. A good example of this is the study of Rosenberry and Winter (1997) which showed that evapotranspiration in an upland area separating two prairie-pothole wetlands in North Dakota (USA), combined with highly variable rainfall, led to the formation of a water table trough between the wetlands in dry years and a water table mound in wet years, and that this dynamic groundwater behaviour affected water levels in the wetlands. Changes in GW-SW interactions over the long-term will occur when there are changes in the heads driven by factors such as climate change, modifications to the management of the uplands/hillslopes (i.e. land use change such as clearing of native vegetation for dryland agriculture, irrigation, forestry, urban development etc.) and/or the riparian zone (urbanisation, agricultural development etc), and/or changes in the flow regimes of the river due to regulation, channelisation, upstream water abstractions etc. For example, Wurster *et al.* (2003) describe an interesting historical case study where over a 100 groundwater-fed interdunal wetlands in Colorado (USA) had disappeared between 1937 and 1995 due to changes in the climate of the region. Conversely, some land use and river management changes can lead to rises in water tables which result in continual movement of groundwater into ephemeral wetlands. The life cycles of many species in arid/semi-arid wetlands require periods of drying which are lost if there is continual movement of groundwater into a wetland. An example of where changes in upland land use and river management has led to rises in riparian zone groundwater levels is the lower Murray River in south-eastern Australia (Walker, 1992; Walker *et al.*, 1992; Walker and Thoms, 1993; Jolly, 1996).

The exchange of groundwater in wetlands is often strongly governed by local geomorphology. Most wetlands occur at low points in the terrain where heavy textured soils are commonplace due to alluvial depositional processes. If the hydraulic conductivity of these clays and silts are lower than that of the underlying aquifer then they can impede groundwater movement between the aquifer and the wetland. While the study of Lamontagne *et al.* (2005) is concerned with a river rather than a wetland, it is a very good example of the role that heavy textured banks/beds can play in controlling GW-SW interactions in semi-arid floodplains. If the soils are very high in sodium (i.e. as a result of natural or human-induced salinisation processes) then the impedance can be further exacerbated because these soils can disperse and swell when wetted with low salinity surface water, leading to significant reductions in hydraulic conductivity (i.e. Jolly *et al.*, 1994).

Wetland and groundwater flow geometry are also important controls on the exchange of groundwater at the wetland scale, as demonstrated by a series of theoretical and field studies by Nield *et al.* (1994), Townley and Davidson (1988), Townley and Trefry (2000), Smith and Townley (2002) and Turner and Townley (2006). These studies have highlighted that GW-SW interactions in wetlands can be broadly classified into three flow regime types (Figure 4): (i) recharge - wetland loses surface water to the underlying aquifer; (ii) discharge - wetland gains water from the underlying aquifer; or (iii) flow through - wetland gains water from the groundwater in some locations and loses it in others. It is important to note that individual wetlands may temporally change from one type to another depending on how the surface water levels in the wetland and the underlying groundwater levels change over time in response to climate, and catchment and river management.

In areas where there are large differences in the salinities of the groundwater and the surface water in the wetland then density-dependent flow processes can be an important control on movement of groundwater into and out of wetlands. Langevin *et al.* (2005) and Jolly *et al.* (1998) are two such examples where this occurs.

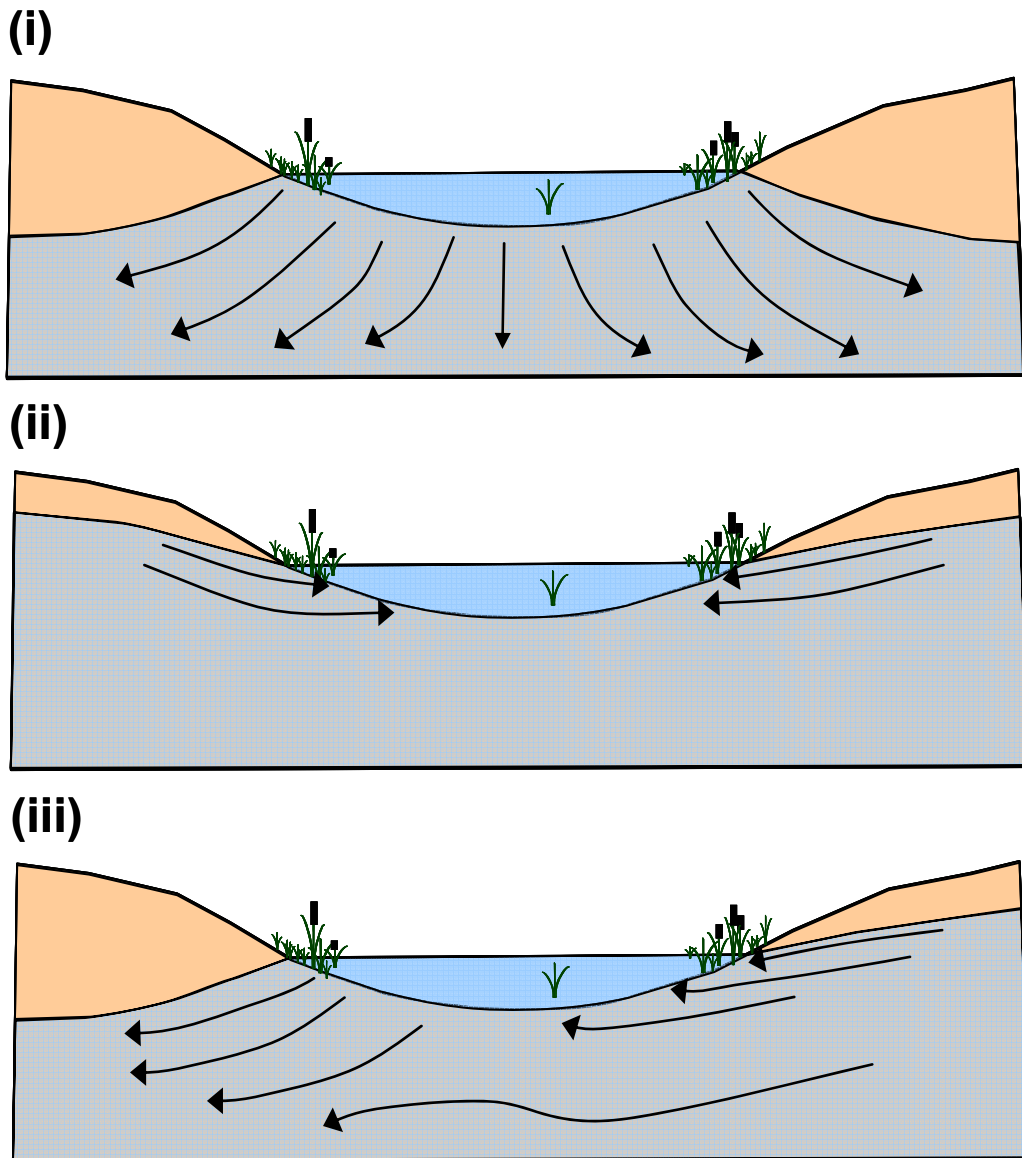


Figure 4. Conceptual groundwater flow paths to and from a (i) recharge; (ii) discharge; or (iii) flow through wetland.

2.3. Wetland Salinisation

In many arid/semi-arid areas there are naturally high concentrations of salts stored in soil and groundwater systems due to factors such as low relief, little or no surface drainage, and high rates of evapotranspiration (Herczeg *et al.*, 2001). The salts can originate from rock weathering, airborne oceanic aerosols transported inland by rainfall (referred to as cyclic salts), connate water trapped in sediments which were deposited in earlier geological times (i.e. from previous sea water transgressions), or aeolian clays (referred to as parna; Butler, 1956). This natural storage of salt in soils and groundwater is referred to as primary salinity.

As described above, land use/management changes in the uplands/riparian zone and/or changes in river management can lead to modifications to groundwater flow regimes. This in turn can lead to the mobilisation of the stored salts which can lead to increased salinisation of soils and surface water bodies such as river, lakes and wetlands. This human-induced movement of salt through the landscape is referred to as secondary salinity and occurs in many arid/semi-arid countries, for example Argentina, Australia, Egypt, Iran, India, Pakistan and South Africa (Ghassemi *et al.*, 1995).

The salinity in wetlands of arid/semi-arid environments will vary naturally due to high evaporative conditions, sporadic rainfall, groundwater inflows, and freshening after rains or floods. However, wetlands are often at particular risk of secondary salinity because of their generally low position in the landscape which exposes them to increased saline groundwater inflows caused by rising water tables (Cramer and Hobbs, 2002). In these situations salt accumulates in the bed of the wetland and in the water column (e.g. Banks, 2003; Baird, 2005) unless there is sufficient low salinity flushing flows. Terminal wetlands are potentially at higher risk than flow through systems as there is no salt removal mechanism (Cramer and Hobbs, 2002). Increased runoff is also a commonly observed consequence of secondary salinity, consequently wetlands can experience increased surface water inflows which are often higher in salinity than under natural conditions. Lakes Toolibin, Towerrinning, and Warden and Coomalbidgup Swamp in southwestern Australia are notable examples of wetlands that have been impacted by secondary salinity (see Froend *et al.*, 1987; Froend *et al.*, 1997; Wallace, 1997; Froend and McComb, 1991; Marimuthu *et al.*, 2005; Froend and van der Moezel, 1994; Davis and Froend, 1999; Halse *et al.*, 2003).

2.4. Wetland Water and Salt Balances

In general, wetlands have historically been described by their surface water regime, which is characterised by water depth through time and relates to the duration of inundation, seasonality, rate of rise, frequency, interflood level and variability (Roberts *et al.*, 2000). The contribution from groundwater is progressively being included into the characterization of the water regime of wetlands. For example Roberts *et al.* (2000) use a water balance approach to describe the water regime of wetlands by including surface water, groundwater, atmospheric water, and the storage volume (as this includes surface water and sub-surface water such that “all the pathways of water movement above and below the ground surface within the wetland are accounted for”). Hayashi and Rosenberry (2002) refer to the hydroperiod of ephemeral wetlands as determined by climatic factors (precipitation and evaporation), amount of surface runoff and input from groundwater exchange. The water and salt balance of a wetland is described by the following (Figure 5):

$$P + S_I + G_I = ET + S_O + G_O + \Delta S \quad (1)$$

$$PC_P + S_I C_{SI} + G_I C_{GI} = S_O C_{SO} + G_O C_{GO} + \Delta S \Delta C_S \quad (2)$$

where P is precipitation, S_I is surface water inflow, G_I is groundwater inflow, ET is evapotranspiration, S_O is surface water outflow, G_O is groundwater outflow, ΔS is change in water storage, C_P is salinity of precipitation, C_{SI} is salinity of surface water inflow, C_{GI} is salinity of groundwater inflow, C_{SO} is salinity of surface water outflow, C_{GO} is salinity of groundwater outflow, and ΔC_S is the change in salinity of the water storage. Note that the salinity of ET is zero as salt is left behind during evaporation.

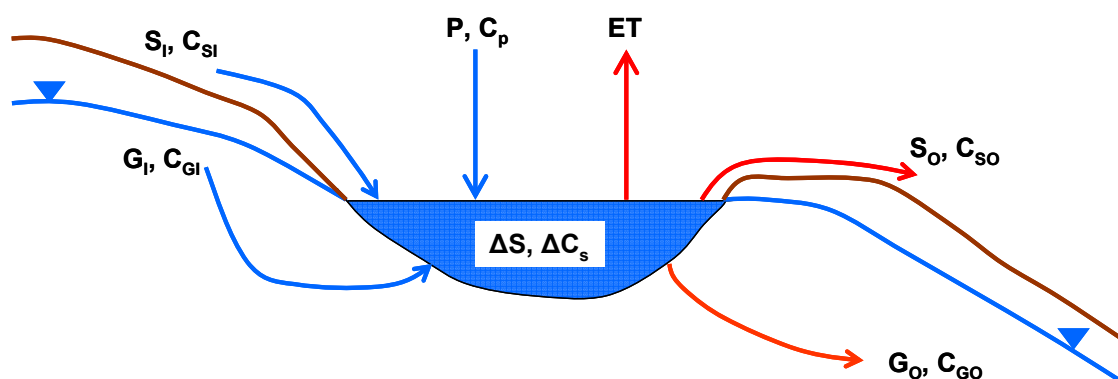


Figure 5. Schematic of the components of the water and salt balance of a wetland.

Groundwater inputs and outputs are generally considered one of the most difficult components of the wetland water balance to characterise because they tend to be very small, compared to surface water inputs and rainfall. (Hunt *et al.* (1996); Hunt *et al.* (1998); and Hunt *et al.* (1999) provide good summaries of methods for measuring groundwater exchange with wetlands). This difficulty in quantifying groundwater inputs and outputs is further complicated by the possible variations in the surface water connection between wetlands and surface water bodies. The majority of literature on the groundwater components of wetland water balances concerns studies in temperate areas (e.g. Gilvear *et al.*, 1993; Rosenberry and Winter, 1997, Hunt *et al.*, 1999; Raisin *et al.*, 1999) and they generally do not consider wetland salt balances. There is a dearth of relevant wetland water and salt balance studies in arid/semi-arid areas.

One study of interest is that of Hayashi *et al.* (1998a) and Hayashi *et al.* (1998b) who studied the transfer of water and chloride between a wetland and its adjacent uplands in a semi-arid area of Saskatchewan, Canada. Their results show that while there was large infiltration of water under the wetland, less than 0.5% of it recharged the underlying aquifer, the rest moved horizontally in the groundwater to the adjacent upland areas where it was transpired by trees and crops. As a result, chloride was found to move from the adjacent upland areas to the wetland by mixing with snowmelt runoff, then infiltrate under the wetland and move laterally back up to the uplands with the shallow horizontal groundwater flow. Under the upland, the chloride moves upward in the vadose zone with soil water, and accumulates near the ground surface as the water is lost by evapotranspiration. A proportion of the chloride mixes with snowmelt runoff and is transported back to the wetland. This chloride cycling was found to occur within 5-6 m of the ground surface.

Marimuthu *et al.* (2005) used geochemical and stable isotope data to supplement traditional hydraulic data in order to unravel the complex GW-SW interaction processes for a series of coastal dune wetlands in a semi-arid area near Esperance, Western Australia. Water quality (particularly salinity) in the wetland system was being affected by land use changes in the surrounding catchment due to volumetric increases in the influx of surface waters and solutes. It was unclear how this was affecting the groundwater inputs to the wetlands from regional aquifers, and whether the effects were occurring uniformly over the eight wetlands that comprised the system. The study found that the water balance of the wetlands system cannot be treated as a single water body, as was perceived from bathymetric data. While a comprehensive water and salt balance was not completed, the study is an interesting one because it illustrates the value in supplementing traditional hydraulic methods with hydrochemistry when investigating complex wetlands systems and their interactions with groundwater.

Although they did not assess salt balances, Wurster *et al.* (2003) demonstrated that the water balance of groundwater-fed wetlands can be dramatically altered by changes in the groundwater conditions in the surrounding areas. Their example was the disappearance of more than 100 interdunal wetlands in the arid Great Sand Dunes National Monument in Colorado, USA. Using a range of hydrological, hydrogeological, hydrochemical, and ecological techniques they concluded that these wetlands were in fact ephemeral features that disappeared for several years during dry periods in which regional water table levels drop. However, the wetlands return during prolonged wet periods when water tables rise and they then may persist for several decades. In addition to the obvious ecological implications for the wetlands themselves, the dry periods also reduce upland vegetation cover which may allow increased dune movement and the partial or complete burial of some wetlands.

In summary, the few available studies suggest that in arid/semi-arid areas the interactions between wetlands and groundwater are highly dynamic, are both temporally and spatially complex, and often extend beyond the surface water boundaries of the wetland. In areas where groundwater is low in salinity, it has beneficial impacts on wetland ecology which can be diminished in dry times when groundwater levels and hence inflows to wetlands are reduced or even cease. Conversely, if groundwater is saline, and inflows increase due to raised groundwater levels caused by factors such as land use change and river regulation,

then this may have detrimental impacts on the ecology of a wetland and its surrounding areas.

2.5. Wetland Groundwater - Surface Water Modelling

Modelling of the interaction of groundwater with shallow water bodies such as lakes has been an active area of research for over 30 years, with seminal early work carried out by Winter (1976), Winter (1978) and Winter (1983). While most of the early studies were site specific, efforts soon turned to developing generic relationships between the geometry of water bodies and the lake-aquifer interactions (e.g. Nield *et al.*, 1994; Townley and Davidson, 1988; Townley and Trefry, 2000). It is not unreasonable to assume that these relationships also hold for wetlands, provided their geometry and field conditions conform to the simplifying assumptions of these approaches. One important criticism that can be made of some of these theoretical studies is that they generally do not adequately consider that fact that most lakes and wetlands contain bottom sediments which have different hydraulic characteristics to the aquifer. While this was taken into account, at least in a simplified manner, in the two dimensional analyses of Nield *et al.* (1994), it has been ignored in subsequent three dimensional analyses such as those of Townley and Trefry (2000) and Smith and Townley (2002). Another shortcoming of these generic approaches is that they assume steady-state conditions, and these rarely hold for most arid/semi-arid wetlands as they typically undergo cycles of wetting and drying resulting in transient GW-SW interactions. In recent years the focus of groundwater-lake interaction modelling has shifted back to site specific studies using traditional groundwater modelling approaches such as MODFLOW (Restrepo *et al.*, 1998; McDonald and Harbough, 2003) and newer analytic element techniques (Haitjema, 1995) and link-node approaches (Walton *et al.*, 1996). Hunt *et al.* (2003) compare the use of the first two of these techniques. It is important to note that all of these studies deal with water flow only.

While the groundwater modelling approaches described above can simulate the hydrogeological complexities beneath wetlands, and provide a means of simulating the transient nature of saturated zone GW-SW interaction, they cannot model the unsaturated zone processes within wetlands (the link-node model of Walton *et al.* (1996) has this capability but was not applied in the case study described in the paper). These are very important in terms of ecological responses, particularly in relation to vegetation growth, decomposition and nutrient release. With these shortcomings in mind, Bradley and Gilvear (2000) modelled the unsaturated zone in and around a degraded floodplain wetland in Leicestershire in England using the one dimensional saturated-unsaturated flow model UNSAT1 (van Genuchten, 1978). The model was used to explore the interaction between water table position, vertical soil moisture fluxes and evapotranspiration. Further development of this type of approach was carried out by Joris and Feyen (2003) who applied a two dimensional saturated-unsaturated flow model (HYDRUS-2D; Šimůnek *et al.*, 1999) to a groundwater fed riparian wetland in Flanders in Belgium. Similarly, Dekker *et al.* (2005) applied HYDRUS-2D to an acidified floating fen in Holland. While the results of these studies are specific to temperate wetlands, and considered only movement of water and not salt, the general approaches should have relevance for wetlands in arid/semi-arid areas.

Another approach is the FEUWAnet model of Dall'O' *et al.* (2001) which simulates the spatial heterogeneity and temporal variation of lateral (from catchment areas to water body) and vertical (from vegetation canopy to groundwater) water fluxes of riparian wetland ecosystems. The model consists of multiple boxes representing various water storage compartments (soil, groundwater, open water etc.) connected by a network of hydraulic resistances. While FEUWAnet was developed for temperate riparian wetlands in northern Germany, for the purpose of assessing the effectiveness of riparian systems in controlling the fluxes of nonpoint pollution discharging to open water bodies, the concepts should also be applicable to arid/semi-arid wetlands, and in principle could be extended to modelling salt movement as well. Other box-type models of wetland-catchment-groundwater movement includes those of Whigham and Young (2001) and Krasnostein and Oldham (2004), although the former does not include interactions with groundwater.

In summary, there has been almost no modelling of GW-SW interactions in arid/semi-arid wetlands with respect to water fluxes, let alone movement of salt. However, in temperate regions there has been a reasonable amount of modelling of water exchange between wetlands, their catchments and groundwater; the studies highlighted above are a sample of this literature. While the results of these studies are generally site specific, the principles and approaches have potential application in arid/semi-arid areas. In addition there is a clear need to develop modelling capabilities for the movement of salt to, from and within wetlands. Box-type modelling is one type approach, another being fully coupled flow and transport modelling along the lines of the recent studies of Bauer (2004), Bauer *et al.* (2006), Langevin *et al.* (2005) and Zimmermann *et al.* (2006).

3. Ecological Responses to Changes in Salinity as a Consequence of the Changes in Surface Water and Groundwater Regimes

As described in Section 1.2.1 climate variability and subsequent fluctuating hydrology are the key drivers of ecology in arid/semi-arid environments. Whilst numerous studies have examined the relationship between wetland biota and water regime (e.g. Casanova and Brock, 2000; Roberts and Marston, 2000; Roberts *et al.*, 2000) there have been few studies of ecological response to both changing water regime and salinity (Brock *et al.*, 2005).

In this chapter the available literature on ecological responses to increases in wetland salinity that occur as a consequence of changes in surface water and groundwater regimes caused by secondary salinity is reviewed. Firstly, the available salinity tolerance literature for wetland biota is summarised, with a discussion of a range of limitations and other issues associated with the use of this data and knowledge. This is followed by a summary of what is currently known of the combined impacts of salinity and changes in water regime on arid/semi-arid wetland ecology. Finally, the impact that changes in GW – SW interactions may have on hydrological pulse events which are key drivers of wetland ecology in arid/semi-arid areas are discussed.

3.1. Available Salinity Tolerance Data

There have been a number of comprehensive summaries and reviews of the salinity tolerance of freshwater aquatic biota and riparian vegetation, and so we do not intend to present individual datasets here. These reviews have been carried out in Australia where secondary salinity is having a significant impact on ecology. Hart *et al.* (1991) concentrated on Australian data and concluded that direct adverse impacts on wetland biota would occur if salinity increased to around 1,000 mg L⁻¹. From salt lake studies, Williams *et al.* (1990) found that the diversity of invertebrates decreased rapidly as salinity increases up to 10,000 mg L⁻¹, but the decline in diversity slowed thereafter. Bailey and James (2000) conducted a review of the international scientific literature pertaining to the impacts of secondary salinity on aquatic systems. The review highlighted a general lack of data worldwide on the sensitivity of freshwater plants and animals to increases in salinity. Niknam and McComb, (2000) reviewed salinity tolerance screening of woody Australian native trees, including several riparian species commonly found fringing arid/semi-arid wetlands. These authors found that there were sometimes conflicting results between glasshouse and/or field trials. The literature review of Clunie *et al.* (2002) focused on work carried out since the review of Bailey and James (2000) and provided an overview of the current knowledge of effects of salinity on freshwater biota and community structure. It concluded that knowledge was still incomplete but was able to make several broad generalisations concerning the effects of salinity on specific taxa, groups of taxa and community structure, mostly focussing on aquatic macrophytes, invertebrates and fish. The review of James *et al.* (2003) discussed what was known of the impacts of secondary salinity on Australian freshwater biota and the resilience of communities within salinising landscapes. One of their conclusions was that the available information needed to be provided to managers in a form that aided their decision making, and as such a large body of data in the form of a salinity-tolerance database was needed. Nielsen *et al.* (2003a) reviewed how salinity affects the physical and biotic components of freshwater aquatic ecosystems and highlighted the need for information on how structure and function of aquatic ecosystems change with increasing salinity. One of the recommendations of James *et al.* (2003) was addressed by the development of the Australian Biodiversity – Salt Sensitivity Database by Bailey *et al.* (2002). This database used presence/absence based on the salinity range and maximum salinity (commonly referred to as the maximum field distribution, MFD) in which a species was observed in the field.

The most recent integration studies have concentrated on the development of ecological risk assessment frameworks and modelling approaches (e.g. Hart *et al.*, 2003; Cant *et al.*, 2003

and Rutherford and Kefford, 2005). In developing their framework Cant *et al.* (2003) summarised the critical salinity thresholds for a number of groups of Australian native taxa and these are reproduced in Table 1.

Table 1. Thresholds of salt sensitivity for groups of taxa of native plants and animals (LC₅₀ is the salt concentration at which 50% of the sample population suffers mortality in laboratory experiments). This is a reproduction of Table 2-3 of Cant *et al.* (2003).

Taxa	Threshold mg L ⁻¹ NaCl	Effect	Source
Macrophytes	1 000 - 2 000	Adverse effects in some species	(Hart <i>et al.</i> , 1991)
Invertebrates	1 000	Adverse effects in some species	(Hart <i>et al.</i> , 1991)
Invertebrates	>800	Adverse effects in some species	(Bailey and James, 2000)
Submerged freshwater macrophytes	1 000 - 2 000	Lethal effects for some species	(Bailey and James, 2000)
Widespread macrophytes	44 000	Disappeared from wetlands	(Brock, 1981); (Brock, 1985); (Brock and Lane, 1983); (Brock and Shiel, 1983); (Garcia, 1999)
Adult freshwater fish	8 800	Adverse effects in some species	(Hart <i>et al.</i> , 1991)
Chara spp.	1 000-3 000	Disappeared from wetlands	(Garcia, 1999)
Nitella spp.	1 000-5 000	Disappeared from wetlands	(Garcia, 1999)
Waterbird broods	15 300	Upper limit for the majority of waterbird broods	(Clunie <i>et al.</i> , 2002)
Frogs	Generally not adapted to saline environments		(Ferraro and Burgin, 1993)
Freshwater tortoises possessing functional salt glands	5 000	Indirect evidence suggests they may be able to cope up to this level	(Hart <i>et al.</i> , 1991)
Acacia spp.	20 000 - 55 100	LC ₅₀	(Morris, 1984);(Morris <i>et al.</i> , 1994)
Banksia spp.	30 000 - 55 100	LC ₅₀	(Morris, 1984);(Morris <i>et al.</i> , 1994)
Callistemon spp.	20 000 - 55 100	LC ₅₀	(Morris, 1984);(Morris <i>et al.</i> , 1994)
<i>E. camaldulensis</i> (River Red Gum)	20 000 - 55 100	LC ₅₀	(Morris, 1984); (Morris <i>et al.</i> , 1994); (Thomson, 1988)
Other eucalypts	19 000 - 55 100	LC ₅₀	(Morris, 1984); (Morris <i>et al.</i> , 1994); (Thomson, 1988)
Grevillea spp.	17 400 - 43 500	LC ₅₀	(Morris, 1984);(Morris <i>et al.</i> , 1994)
<i>Maleleuca</i> spp.	23000 - 55 000	LC ₅₀	(Morris, 1984);(Morris <i>et al.</i> , 1994)

3.2. Knowledge Gaps and Other Issues Associated With the Available Salinity Tolerance Data

The above reviews have highlighted a number of key gaps and problems with the available salinity tolerance data and understanding, including:

- there is little or no information on the salt sensitivity of micro-algae, frogs, platypus and tortoises,
- there are question marks as to how representative controlled glasshouse and laboratory studies are of the highly variable and unpredictable conditions experienced in the field,
- there are deficiencies in using point based salinity measurements and presence/absence data due to the high spatial and temporal variation in salinities that occur in freshwater systems. There is often potential for false absences and high sampling biases because of uneven numbers of observations across salinity categories,
- that surface water salinity measurements are not indicative of soil and groundwater conditions experienced by emergent and deep rooted vegetation,
- most studies have only sampled adults or juveniles and so there is an inadequate understanding of sub-lethal effects and sensitive life stages,
- the results do not account for the fact that salinity also has indirect effects on pH, dissolved oxygen and nutrient balances of plants, which may be as significant as the direct ion effect of salt alone.,
- many studies only report salinity thresholds such as MFD rather than growth changes with increasing salinity and lengths of time that salinity levels remain elevated,
- the inclusion of data from estuarine and marine systems can be confusing when using the databases to evaluate land-locked wetlands that were initially fresh,
- there is a limited understanding of the impacts of salinity on species' interactions, food web structures, and the structure and integrity of communities,
- the interactions between salinity and other environmental stressors and perturbations such as water regime, waterlogging, eutrophication, turbidity, pH etc have been rarely studied in naturally occurring wetlands (as opposed to constructed wetlands), and
- there is very little time series data which is a serious issue given that wetland salinities are often highly temporally variable due to changes in surface water inflows and outflows, evapoconcentration, and fluctuations in groundwater inputs and losses.

Some of these issues are further addressed below, along with research results that have been published since the last of the above reviews.

3.2.1. Field Versus Laboratory Studies of Salinity Tolerance

In theory, laboratory and glasshouse studies can be used to unravel the highly complex interactions between ecology and changing water regime and salinity regimes. However, it is now becoming clear that these types of studies, in which water, salinity, nutrition and energy regimes are often kept constant or are closely controlled, are not necessarily representative of the highly variable and unpredictable conditions experienced in the field (Jolly *et al.*, 2002).

For riparian vegetation that fringes wetlands most of our knowledge on the salinity tolerance of species is derived from laboratory and glasshouse studies (Niknam and McComb, 2000). One of the problems in utilizing and comparing data from these studies is the lack of

uniformity in the treatments tested (differing salinity levels, lengths of waterlogging, study durations, response measurements etc.). Moreover, in recent times it has become clear that these types of studies, in which the salinity, irrigation, nutrition and energy are generally kept constant or are closely controlled, are often not representative of the highly variable conditions experienced in the field. Furthermore, the majority of these studies use individual juvenile plants (up to 1 year old), and so the observed responses are unlikely to be representative of mature communities of plants. Also, Bailey *et al.* (2002) point out that surface water salinities can often differ considerably to both soil and groundwater salinities. Thus aquatic and emergent vegetation which inhabit wetland sediments or hyporheic zones may experience quite different salinities to those indicated by the surface waters. Deeply rooted riparian vegetation may avoid high soil salinity by utilising groundwater where it is less saline. It is difficult to accurately replicate these complex field processes in glasshouse and laboratory experiments.

In the case of emergent and submerged or floating aquatic vegetation, Bailey *et al.* (2002) are of the view that glasshouse and laboratory experiments can provide an indication of sublethal effects of salinity on vegetation. In these studies it has been demonstrated that profound growth reductions can occur at considerably lower salinities to those which result in mortality. The sublethal responses to salinity are often characterised by reductions in plant biomass, height, flowering, shoot number, leaf number and leaf size. These effects may be apparent at salinities as low as 1,000 mg L⁻¹ (James and Hart, 1993). Leaf production is often slowed, and in some cases premature leaf senescence is induced. Visual symptoms of salt stress include leaf burn, wilting or chlorosis.

In the case of aquatic animals, Kefford *et al.* (2004) recently carried out an experiment where they compared laboratory-derived acute salinity tolerance (the concentration that is lethal to 50% of individuals; LC₅₀) of freshwater macroinvertebrates and fish from south-eastern Australia with the maximum salinity at which they have been recorded alive in nature (i.e. their MFD). Their study found that the MFD of freshwater macroinvertebrate taxa were correlated with their acute (72 hour) direct transfer LC₅₀ salinity values. The correlation was best for common macroinvertebrates because the high number of field observations led to good estimates of MFD. Relative to macroinvertebrates, direct transfer LC₅₀ values from both adult and early life-stage freshwater fish were found to provide a poorer estimate of the MFD values. However, slow transfer (slow increase in salinity over several days to allow fish to acclimatise) LC₅₀ provided a better estimate of the MFD for adult freshwater fish. It was thought that because salinity changes in field situations are often slow, short-lived species may not experience large changes during their life, whereas longer lived and more mobile species may be able to acclimatise and survive in areas where salinity levels would be lethal in their early life stages.

In an effort to improve upon the simple threshold-based approaches, Cant *et al.* (2003) have recently developed an ecological risk assessment framework they refer to as the quantitative Salt Impact Model. The purpose of this approach is to identify the likely impact of salinity on biodiversity assets for which there is little or no toxicity data. It utilises the field-based salinity thresholds in the Salt Sensitivity Database of Bailey *et al.* (2002) where they exist. However, for assets that do not occur in the database salinity effects are estimated based on "vital attributes", characteristics of species or communities that influence their response to a stressor such as salinity (i.e. osmoregulatory structures, dormancy, soil type etc.). While Cant *et al.* (2003) present three case studies illustrating the use of this new approach it is yet to be independently tested in the literature.

3.2.2. Acclimatisation of Biota to Salinity

Acclimatisation to salinity has been demonstrated to enhance salinity tolerance in trees (Heth and MacRae, 1993). However, Clunie *et al.* (2002) highlight that there is very little understanding of the capacity of freshwater biota to acclimatise to salinity, but there is evidence to suggest that recent acclimation history can greatly influence salt sensitivity. Rutherford and Kefford (2005) state that there is evidence that tolerance of a given species may change depending in the history of its exposure, *viz* an individual may adapt to prevailing conditions (Ryan and Davies, 1996), such that salinity tolerance values may not be

transferable from one locality or time period to another. Furthermore, Rutherford and Kefford (2005) found that several species of macroinvertebrates were found at higher salinities in Western Australia than in Victoria and South Australia. They suggest that the more extreme salinity conditions in Western Australia could lead to higher tolerance if macroinvertebrates become adapted to long term exposure to salinity. Pinder *et al.* (2005) also came to the conclusion that the freshwater invertebrate fauna in the wheatbelt of Western Australia may be comparatively salt tolerant, and that this tolerance provides a buffer against the effects of mild salinisation.

3.2.3. Resilience of Biota to Salinity

Resilience is the capacity of a system damaged by disturbance to return to its prior state if the disturbance is removed. Many wetland biota have some natural resilience and adaptive strategies to cope with salinity and dynamic hydrology. As noted by Hart *et al.* (2003), the resilience of most freshwater biota is a function of the use and availability of refugia. They give the example that with salinisation of wetlands, biota such as insects may leave for less saline wetlands and then recolonise the affected wetland later if the salinity drops (i.e. returns to its prior state). If however, the salinisation of the wetland persists, then the system does not recover and the original species are lost and replaced by a reduced set of more tolerant species (i.e. moves permanently to an alternative state). Hart *et al.* (2003) argue that while resilience is a useful theoretical concept, there is currently insufficient knowledge to measure a system's resilience and to use this to set thresholds for particular disturbances. However, they believe that this knowledge is beginning to be generated, the study of Davis *et al.* (2003) being one example. Another recent example is the study of Strehlow *et al.* (2005) which found evidence that the onset of secondary salinity may decrease the resilience of macrophyte dominated systems over time and drive them towards benthic community-dominated systems.

3.2.4. Life Stages and Salinity Tolerance

As summarised by Clunie *et al.* (2002) and Nielsen *et al.* (2003a), our understanding of sublethal effects and sensitive life stages is very poor. The limited evidence available suggests that juvenile life stages are more sensitive than the adults and that the reproductive capacity of adults may be impaired by elevated, but sublethal, salinity levels. Presence of a species at a particular salinity does not necessarily indicate that it can complete its life-cycle at that salinity (Kefford *et al.*, 2004). This has important implications for the long-term structure, function and sustainability of populations and communities, in that whilst elevated salinity levels may not be lethal for adults they may not be able to reproduce and/or there is no recruitment of young to the population. For aquatic biota, Skinner *et al.* (2001), Nielsen *et al.* (2003a), Brock *et al.* (2003) and Brock *et al.* (2005) point out that dormant life stages, such as eggs, seeds, spores and asexual propagules of aquatic organisms are important reservoirs of biodiversity in semi-arid areas, and the juvenile zooplankton and seedling that emerge from this bank (reservoir) may be more sensitive than adult life stages to increasing salinity. Emergence from seed banks has been used to detect change in biotic communities in studies of drought, water regime and salinity in wetlands (Casanova and Brock, 2000; Brock *et al.*, 2003; Nielsen *et al.*, 2003b; Baldwin *et al.*, 2005; Nielsen *et al.*, 2003a).

3.2.5. Transferability of Results

There are questions as to whether the results of an individual study are transferable to other locations and times given the high spatial and temporal variability of hydrological and water quality parameters than can influence wetland health (see Reid and Brooks, 2000). For example, Brock *et al.* (2005) found that the effects of salinity and hydrology on aquatic plant germination and emergence of zooplankton eggs were not similar for wetlands of a wide geographic range within the Murray-Darling Basin in south-eastern Australia. As described above, Rutherford and Kefford (2005) found that several species of macroinvertebrates were found at higher salinities in Western Australia than in Victoria and South Australia.

3.2.6. Interactions Between Salinity and Other Environmental Stressors

As discussed in Clunie *et al.* (2002), the interactions between salinity and other environmental stressors and perturbations such as changed water regimes, eutrophication, turbidity, temperature, pH etc have been rarely studied. One example where this has been attempted is the study of Timms and Boulton (2001) which showed the interplay between salinity, turbidity and water regime in determining invertebrate composition in arid-zone wetlands. In particular they found that whilst species richness varied amongst wetlands, it was assemblage composition that more clearly differentiated wetlands types. This was despite the variable responses to salinity, turbidity and water regime that different taxa exhibited. They concluded that it was not just salinity that was dictating thresholds of tolerance that distinguish different assemblages.

Another recent study which highlighted complex interactions between ecology, salinity, water regime and turbidity is that of Strehlow *et al.* (2005). They carried out detailed time-series sampling of water depth, water quality, submerged macrophyte biomass and macroinvertebrate richness and abundance in 6 wetlands in southern Western Australia over an 18 month period which encompassed a complete wetting and drying cycle. They found that high turbidity occurred when salinity was highest due to wind-induced sediment re-suspension in shallow water. They also found that very shallow wetlands were able to support aquatic plant communities and suggested that depth may be affecting these plant communities indirectly by regulating the salinity and turbidity. The responses of invertebrate communities over time varied from site to site, and the differences in the species present at each wetland suggested there was high specificity of fauna in response to the prevailing salinity and hydrological regime. They concluded that a change in ecological regime may be a more important threshold for determining the impacts of secondary salinity on ecology, rather than the increase in salinity alone.

3.3. Combined Impacts of Salinity and Changes in Water Regime

The onset of secondary salinity not only increases in the salinity of wetlands, but can also impact on their water regime through increased groundwater inflows which increase water depths and extend lengths of inundation (i.e increases hydroperiod) (Cramer and Hobbs, 2002). Bailey *et al.* (2002) hypothesised that it can be anticipated that since salinity reduces plant height that increases in water depth will compound growth reductions imposed by salinity. Furthermore, increased water depths can result in water logging of fringing vegetation, and as pointed out by Clunie *et al.* (2002), increasing salinity and water logging can act synergistically in affecting plant growth. In this section we summarise a number of studies where the combined effects of changes in surface water regime and salinity have impacted on wetland ecology.

A large number of intensive field and modelling studies on the floodplains of the lower River Murray in Australia have examined the interaction between groundwater, surface water, salt accumulation, water use and growth of riparian vegetation (*Eucalyptus camaldulensis* and *E. largiflorens*) which commonly fringe arid/semi-arid wetlands in south-eastern Australia (e.g. Jolly *et al.*, 1993; Thorburn *et al.*, 1993; Thorburn and Walker, 1993; Mensforth *et al.*, 1994; Thorburn and Walker, 1994a; Thorburn and Walker, 1994b; Thorburn *et al.*, 1995; Jolly and Walker, 1996; Taylor *et al.*, 1996; Jolly, 1996; Akeroyd *et al.*, 1998; Slavich *et al.*, 1999a; Slavich *et al.*, 1999b; Holland *et al.*, 2006; Overton *et al.*, 2006; Doble *et al.*, 2006). Accumulation of salt in floodplain soils occurs naturally in the lower River Murray region, and prior to river regulation was mitigated by leaching from frequent floods which inundated the floodplains. Over the long-term, there was a salt balance in the soil that enabled the development of long-term stable vegetation communities, as evidenced by measurements on Black box trees on the Chowilla floodplain that show they can live as long as 450 years (Slavich, 1997). However, over the last 70 to 80 years, the surface and groundwater hydrology of the floodplains has been dramatically altered due to the impacts of river regulation and the development of large irrigation areas on the higher areas adjacent to the

floodplains. Salt is now accumulating in floodplain soils at an increased rate due to increased groundwater discharge caused by raised water table levels and point source saline discharge from adjacent irrigation areas. These problems are further exacerbated by less frequent leaching of the accumulated salt in the soils by large floods, the frequency and duration of which has been greatly reduced by river regulation. In addition to having severe impacts on the health of the adult populations of these species, the high soil salinity and lack of flooding is greatly affecting recruitment (George *et al.*, 2005).

In the Upper South-East region of South Australia, Mensforth (1996) and Mensforth and Walker (1996) studied the growth of another species commonly found fringing arid/semi-arid wetlands in Australia (*Melaleuca halmaturorum*). This study highlighted how this species can dynamically alter its water use and source throughout time in response to fluctuating saline shallow groundwater and surface water availability. They found that these trees used groundwater from the soil surface at the end of winter in response to groundwater rise and inundation of the soil profile. During summer they used water from deeper in the soil profile in response to salt accumulation in the surface soils. At the end of summer, although there were high salt concentrations near the soil surface the soil was generally moist and roots were extracting water from below the saline zone. Winter rainfall quickly recharged the groundwater, leached salt from the soil profile, and caused the moderately saline groundwater to rise to the soil surface. Waterlogging caused the roots to die back to near the soil surface. Subsequent evapotranspiration then caused the water table to drop quickly. The roots began to grow, drying the soil until the concentration of salts in the soil solution was too high for more water to be extracted. The roots once again began to die back at the soil surface due to the high salinity. This sequence of events occurred over a time period of 12 months (Figure 6).

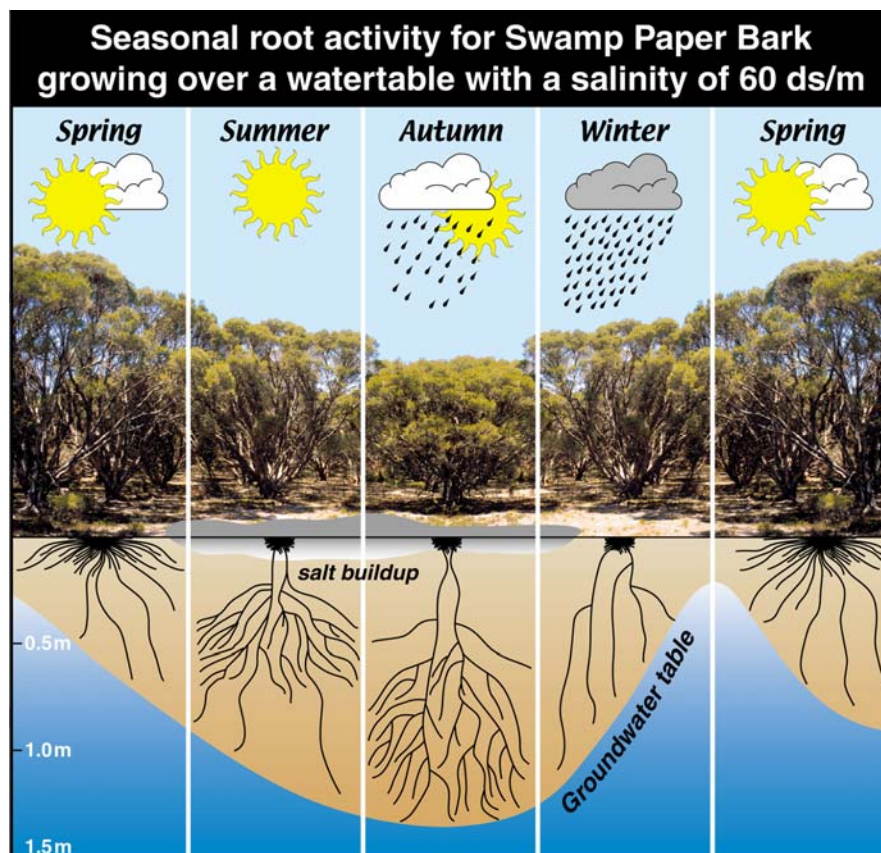


Figure 6. Seasonal root activity and depth of water extraction of *Melaleuca halmaturorum*. From Jolly *et al.* (2002) based on the study of Mensforth (1996) and Mensforth and Walker (1996).

Davis *et al.* (2003) recently pointed out that as secondary salinity generally occurs as a result of rising water tables, the most immediate impact on wetland ecology is often increased water depth and loss of seasonal wetting and drying cycles, rather than the effects of higher salinity alone. They cite the oral histories recorded by Sanders (1991) which suggest that in the wheatbelt wetlands of Western Australia, fringing, emergent and freshwater aquatic vegetation dies within ~5 years of the onset of secondary salinity, often from waterlogging rather than salt *per se*. They also cite Halse *et al.* (1993b) who believe that secondary salinity in this region has changed wetland plant communities from those dominated by woody species to those dominated by submerged species, with major consequences for fauna reliant on emergent plants, particularly waterbirds. Davis *et al.* (2003) provide the case example of Toolibin Lake in the south-west of Western Australia, in which permanent inundation over prolonged time periods has resulted in the death of wetland tree communities and the prevention of recruitment. Toolibin Lake is the last remaining freshwater wetland in the region with extensive stands of *Casuarina obesa*, *Melaleuca* spp. and *Eucalyptus rudis* (Froend *et al.*, 1987). This vegetation association is typical of the naturally occurring wetland vegetation of this area prior to rises in groundwater induced by clearing for agriculture (Halse *et al.*, 1993a). Measurements of soil salinity and the calculation of percentage inundation from tree elevation and observations of tree vigour and xylem pressure potential response indicated that tree deaths in the *Melaleuca* spp. and *C. obesa* were due to increased levels of salinity. Death and low vigour in *E. rudis* was attributed to both increasing salinity and prolonged inundation (Froend *et al.*, 1987). Secondary salinisation has had a greater effect on the lake margin species than on those inhabiting the environments of the lake bottom or the uplands region, which are unaffected by inundation. The lake is important as a breeding area for waterbirds as extensive dense thickets of *C. obesa* and *Melaleuca* spp. occur through much of the inundated area. Live vegetation in the lake is important for providing suitable nesting sites and the fresh/brackish water is of sufficiently high quality for growth of emergent vegetation and is suitable for young birds. The periodic drying of the lake also allows persistence of the trees growing on the lake bed. Toolibin Lake has been the subject of remediation including short to medium term engineering measures to decrease salinity within the lake and long-term rehabilitation measures within the catchment (Froend *et al.*, 1997; Wallace, 1997)(Figure 7). Another wetland in the south-west of Western Australia that has been affected by secondary salinity and changes in water regime is Lake Towerrinning. By the late 1980s it was highly salinised and eutrophic, had lost its fringing vegetation and contained little aquatic life. (Froend and McComb, 1991). It also has been the subject of engineering works aimed at reducing salinity levels (George *et al.*, 1996).

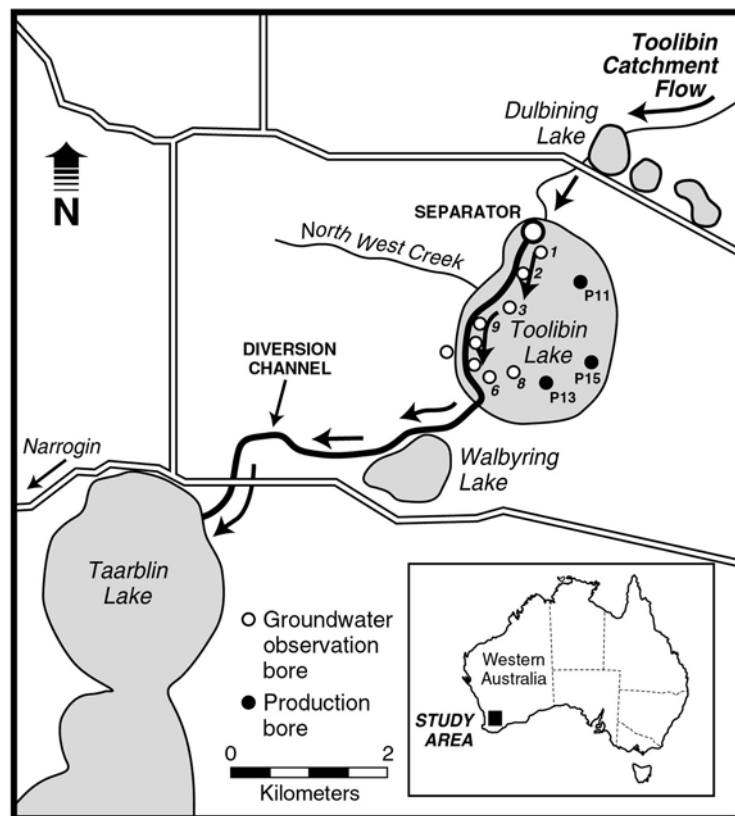


Figure 7. Location of Toolibin Lake and schematic showing diversion channel (to prevent saline surface runoff entering the lake), production bores (to lower groundwater beneath the lake bed) and observation bores. From Jolly *et al.* (2002).

Brock *et al.* (2005) conducted a laboratory study of the effects of salinity and water regime on aquatic biota in sediments from seven wetlands in the Murray-Darling Basin in south-eastern Australia. Their experiments used germination of aquatic plant seeds at five salinity levels (<300, 1,000, 2,000, 3,000, 5,000 mg L⁻¹) and two water levels (damp (waterlogged) and flooded). They also studied the emergence of zooplankton eggs at the same five salinity levels. For sediments from four of the wetlands, they found that there was decreasing species richness and abundance of biota germinating or hatching as salinity increased above 1,000 mg L⁻¹, and that this trend was more marked when sediments were damp rather than flooded. For the other three wetlands results were more variable. As discussed in Section 3.2.5, the lack of consistency of responses raises questions about transferability of results between wetlands. Notwithstanding this issue, they concluded that there was greater loss of diversity for plants germinating from seed banks at the edges of wetlands where plants are not completely submerged than for those in submerged conditions. They suggested that transpiration and evaporation at the exposed soil surface and ion exclusion mechanisms in plant roots could cause localized increases in salinity in the root zone under damp conditions (i.e. the fringe of the wetland), whereas in flooded conditions (i.e. the wetland itself), constant submergence could prevent or ameliorate accumulation of salts. Overall, Brock *et al.* (2005) concluded that their “results suggest that the effects of salinity on plants germinating from a seed bank will be more pronounced at wetland edges or in temporary wetlands where water levels fluctuate, than from the same seed bank germinating in permanently flooded areas of wetlands”. A possible implication of this is that if secondary salinity also leads to greater wetland water depths then these may in part offset the impacts of the increasing salinity on aquatic vegetation. Clearly further studies are required to determine if this is indeed the case and whether the results of Brock *et al.* (2005) apply more generally than just the small number of wetlands they studied.

In summary, it is clear that secondary salinity can impact on wetland biota through changes in both salinity and water regime which result from the hydrological changes caused by secondary salinity. Whilst there have been some detailed studies of these interactions for

some Australian riparian tree species, as indicated by Brock *et al.* (2005), the combined effects on aquatic biodiversity have not yet been fully elucidated.

3.4. Pulse Events and Groundwater – Surface Water Interactions

Climate variability and subsequent fluctuating hydrology are key drivers of biogeochemical cycles and ecology in arid/semi-arid environments (Austin *et al.*, 2004; Chesson *et al.*, 2004; Huxman *et al.*, 2004; Schwinning *et al.*, 2004; Schwinning and Sala, 2004). Rainfall and flow pulse events drive hydrological fluctuations in wetlands which in turn lead to physiological, morphological, and life-history traits that facilitate survival and growth of biota in the water-limited variable environments of arid/semi-arid regions. For example, in the case of vegetation in arid areas there is a widely cited paradigm, the 'pulse-reserve' conceptual model that depicts a direct relationship between rainfall, which triggers pulses of plant growth, and reserves of carbon and energy (Reynolds *et al.*, 2004). The pulse paradigm has also been applied to rivers and floodplains by Junk *et al.* (1989) who developed the flood pulse concept based mainly on large tropical lowland systems with predictable overbank flood pulses of long duration. This concept was then extended by Tockner *et al.* (2000) to encompass temperate floodplains located anywhere along a river corridor, and to include floodplain expansion-contraction cycles occurring below bankfull (i.e. 'flow pulse' versus 'flood pulse'). Walker *et al.* (1995) discussed the shortcomings of the flood pulse concept in relation to lowland rivers in arid/semi-arid areas (the concept is not easily applied where the pulse is variable, and does not account for the effects of river regulation), but concluded that the concept could be adapted to these environments. Puckridge *et al.* (1998) also concluded that the flood pulse concept could also be extended to encompass the complexity and diversity of hydrological patterns in large rivers in arid zones. Amoros and Bornette (2002) highlight the role of lateral (river to wetlands) and vertical (GW-SW interactions) connectivity in relation to hydrological pulsing that drives the functioning of floodplain ecosystems. From these and many other studies it is clear that the timing and duration of river flow pulses are major factors responsible for composition, structure and function of floodplain wetlands communities. Indeed, the life cycles of many wetland biota are directly related to flow pulses in terms of timing, duration, rise and fall of flood waters (e.g. fish breeding cycles; see Bayley, 1991). Clearly, changes in climate and land management, and regulation of rivers will have significant impacts on rainfall/flow pulse regimes and hence wetland ecology.

In addition to rainfall/flow pulses directly controlling ecological function they also have an important indirect role through their impacts on wetland salinity. Freshwater pulses can be the primary means by which salt stored in both the water column and in the underlying sediments are flushed from wetlands. Conversely, as discussed in Section 2.3, increased runoff is also a commonly observed consequence of secondary salinity and wetlands can experience increased surface water inflows which are often higher in salinity than under natural conditions. Moreover, changes in rainfall/flow pulse regimes can have significant impacts on wetland GW-SW interactions. Because groundwater is generally the primary source of salt in wetlands, and because this is in part controlled by head differences between the surface water in a wetland and the underlying groundwater, any changes in wetland surface water regimes due to changes in rainfall/flow pulsing regimes will also have salinity implications for wetlands. This is even more so the case in wetlands experiencing secondary salinisation as water tables underlying wetlands are rising over time (Cramer and Hobbs, 2002), further increasing groundwater inflows, and hence movement of salt, into wetlands. It is possible that in some instances that the groundwater inflow to a wetland may become so large that it could become a major component of the water balance (e.g. Raisin *et al.*, 1999) and hence mask the role of natural pulsing regimes. On the positive side, if the groundwater is low in salinity it may provide an ecological benefit in arid/semi-arid areas by assisting in maintaining water in wetlands that become aquatic refugia between flow pulses (Hamilton *et al.*, 2005).

4. Conclusions

1. The few available studies suggest that in arid/semi-arid areas GW-SW interactions in wetlands are highly dynamic, are both temporally and spatially complex, and often extend beyond the surface water boundaries of the wetland. In areas where groundwater is low in salinity, it has beneficial impacts on wetland ecology which can be diminished in dry times when groundwater levels and hence inflows to wetlands are reduced or even cease. Conversely, if groundwater is saline, and inflows increase due to raised groundwater levels caused by factors such as land use change and river regulation, then this may have detrimental impacts on the ecology of a wetland and its surrounding areas.
2. GW-SW interactions in wetlands are mostly controlled by factors such as differences in head between the wetland surface water and groundwater (which in turn are controlled by climate, catchment land use and river management), the local geomorphology of the wetland (in particular the texture and chemistry of the wetland bed and banks), and the wetland and groundwater flow geometry.
3. GW-SW interactions in wetlands can be broadly classified into three flow regime types: (i) recharge - wetland loses surface water to the underlying aquifer; (ii) discharge - wetland gains water from the underlying aquifer; or (iii) flow through - wetland gains water from the groundwater in some locations and loses it in others. However, it is important to note that individual wetlands may temporally change from one type to another depending on how the surface water levels in the wetland and the underlying groundwater levels change over time in response to climate, and catchment and river management.
4. The salinity of wetlands in arid/semi-arid environments will vary naturally due to high evaporative conditions, sporadic rainfall, groundwater inflows, and freshening after rains or floods. However, wetlands are often at particular risk of secondary salinity because of their generally low position in the landscape which exposes them to increased saline groundwater inflows caused by rising water tables. Terminal wetlands are potentially at higher risk than flow through systems as there is no salt removal mechanism.
5. There has been almost no modelling of GW-SW interactions in arid/semi-arid wetlands with respect to water fluxes, let alone salinity or ecology. There is a clear need to develop modelling capabilities for the movement of salt to, from and within wetlands so to provide temporal predictions of wetland salinity which can be used assess ecosystem outcomes.
6. There has been a concerted effort in Australia to collect and collate data on the salinity tolerance/sensitivity of freshwater aquatic biota and riparian vegetation. There are many shortcomings and knowledge gaps in these data, a fact recognised by many of the authors of this work. Particularly notable is that there is very little time series data. This is a serious issue given that wetland salinities are often highly temporally variable due to changes in surface water inflows and outflows, evapoconcentration, and fluctuations in groundwater inputs and losses. There is also a concern that many of the data are from highly controlled laboratory experiments which may not represent the highly variable and unpredictable conditions experienced in the field. In light of these, and many other shortcomings identified in Section 3.2, our view is that the data are a useful guide but must be utilised with some caution.
7. Secondary salinity can impact on wetland biota through changes in both salinity and water regime which result from the hydrological changes associated with secondary salinity. Whilst there have been some detailed studies of these interactions for some Australian riparian tree species, the combined effects on aquatic biodiversity have not yet been fully elucidated, and therefore a future research need.

8. Rainfall/flow pulses are a strong control on ecological function in arid/semi-arid areas. They also have an important indirect role through their impacts on wetland salinity. Freshwater pulses can be the primary means by which salt stored in both the water column and in the underlying sediments are flushed from wetlands. Conversely, increased runoff is also a commonly observed consequence of secondary salinity and so wetlands can experience increased surface water inflows which are often higher in salinity than under natural conditions. Moreover, changes in rainfall/flow pulse regimes can have significant impacts on wetland GW-SW interactions. It is possible that in some instances that the groundwater inflow to a wetland may become so large that it could become a major component of the water balance and hence mask the role of natural pulsing regimes. However, if the groundwater is low in salinity it may provide an ecological benefit in arid/semi-arid areas by assisting in maintaining water in wetlands that become aquatic refugia between flow pulses.

Glossary

Note: Many of the following terms are not strict scientific definitions, being rather more colloquial explanations of their meaning and significance.

Anoxia: lack of oxygen in the soil due to waterlogging.

Aquifer: a saturated permeable soil or geologic strata that can transmit significant quantities of groundwater under a hydraulic gradient.

Capillary rise: the upward movement of groundwater through the soil caused by the surface tension of water in soil pores. When water tables rise to near the surface, drying of soil by evaporation and transpiration leads to high rates of capillary rise of groundwater. This is an important form of groundwater discharge and if the groundwater is saline then this process leads to salt accumulation in the soil profile.

Catchment: an area of land bounded by topographic or geologic features within which rainfall and/or groundwater drains to a particular point such as a stream or lake.

Dynamic equilibrium: refers to how a groundwater system/soil hydrology responds to naturally varying rainfall rather than any land use change. Land use change usually results in a transformation to a new dynamic equilibrium.

Ecohydrology: a sub-discipline of hydrology that focuses on ecological processes involved in the hydrological cycle.

Evapoconcentration: concentration of a solute in water due to evapotranspiration.

Evapotranspiration: loss of water to the atmosphere by evaporation from the soil surface and by transpiration from plants.

Groundwater: sub-surface water in soils and geologic strata that have all of their pore space filled with water (i.e. are saturated).

Groundwater discharge: the loss of water from an aquifer (i) to the atmosphere by evaporation, springs and/or transpiration, or (ii) to a surface water body (in the case of rivers it is generally referred to as base flow) or the ocean, or (iii) by extraction.

Groundwater recharge: the addition of water to an aquifer, most commonly through infiltration of a portion of rainfall, surface water or irrigation water that moves down beyond the plant root zone to an aquifer.

Halophyte: plants that grow naturally in high concentrations of salt, and require some salts for optimum growth.

Hydraulic gradient: the change in hydraulic head in an aquifer with either horizontal or vertical distance, in the direction of groundwater flow.

Hydroperiod: the duration, frequency, depth and season of flooding.

Hyporheic zone: the saturated zone of mixing between groundwater and surface water beneath surface water bodies.

Indigenous terrestrial vegetation: naturally occurring native vegetation growing on the land surface (i.e. not in water bodies such as streams, lakes and wetlands).

Ion toxicity: adverse effects of specific ions of salt in a plant that has difficulty in regulating the amount and type of salt taken up by the roots.

Isotope: atoms with the same number of protons but different numbers of neutrons. The slight difference in mass changes some of their physical and chemical properties. Measurement of the ratio of the concentration of the stable isotopes of water and carbon (^2H , ^{18}O , ^{13}C) to their normal form (^1H , ^{16}O , ^{12}C) can be a useful means of tracing sources of water. Use of isotopes that radioactively decay such as ^3H and ^{14}C can be a means of determining the age of water or plant material.

Isotopic signature: naturally occurring ratios of stable isotopes to normal elements in plant and animal tissue, soils and water.

Leaching: the removal of salt from the soil profile by water applied at or near the soil surface (rainfall, floodwater, irrigation). This water drains down through the soil profile taking salt with it.

Leaf area index: the ratio of total single-sided leaf surface area to the area of the canopy projected onto the ground.

Leaf or plant water potential: a measurement that provides an indication of the water status of a plant. It is reported as a negative number with the minimum observed value providing a measure of the plants ability to extract water from soils under extreme drought and/or saline conditions.

Osmotic effects: plant stress and dieback due to physiological drought. Even though adequate water may be available in the soil profile, high concentrations of salt affect the ability of the plant to extract water from the soil (the osmotic potential is too low for the water to be available to the plant).

Physiological adaptations: adaptations as a result of climatic, edaphic, and biotic factors acting on an organism that result in it being better suited to its environment.

Primary salinity: natural soil and/or water salinisation resulting from a limited capacity to drain salt and water from the landscape.

Root zone: the part of the soil profile where plant roots are active.

Salinity threshold: salinity level at which plant growth or health is inhibited.

Salt accumulation: when saline water tables rise near to the soil surface evaporation and water uptake by plants leads to a build up of salts left behind in the soil profile.

Salt balance: when the mass of salt entering a soil profile, groundwater system or catchment is balanced by an equivalent mass of salt exiting the area over the same time period. An example is a catchment in which the mass of salt entering the catchment via rainfall is equal to that leaving the catchment by stream flow at its outlet.

Sap water: water moving in the plant xylem from the roots to the leaves in response to transpiration.

Secondary salinity: soil and/or water salinisation caused by human-induced activities such as land use change.

Soil matric suction or potential: the amount of energy required to remove water held in soil pores due to the forces of surface tension. Can be reported as either a positive (suction) or negative (potential) number and is a measure of the dryness of the soil that is independent of soil texture.

Soil osmotic suction or potential: the amount of energy required to move water through a membrane that is permeable to water but not to solutes (in this context the membrane is a root surface). Can be reported as either a positive (suction) or negative (potential) number and is a measure of dryness of the soil to plants due to salinity.

Surface water regime: a regime that is described as water depth through time and relates to the duration of inundation, seasonality, rate of rise, frequency, inter-flood level and variability.

Transpiration: loss of water vapour and other gases from leaves and other plant surfaces to the atmosphere. Also referred to as plant water use or uptake.

Vadose zone: The zone between land surface and the water table within which the moisture content is less than saturation (except in the capillary fringe) and pressure is less than atmospheric. Also referred to as the unsaturated zone.

Water balance: a state of equilibrium where rainfall or irrigation water in a soil profile, groundwater system or catchment is accounted for by the sum of run-off, transpiration, evaporation, recharge and changes in soil moisture content.

Waterlogging: when a soil profile becomes saturated by either inundation or rising water tables entry of oxygen into the soil is limited. Once all of the existing oxygen in the soil has been used, plants will begin to dieback due to the anoxic conditions, as water uptake by plant roots requires adequate oxygen. Some plant species have adaptations that enable them to better survive such saturated conditions.

Water table: the level of groundwater in an unconfined aquifer. The soil pores and geologic strata below the water table are saturated with water.

References

- Akeroyd, M.D., Tyerman, S.D., Walker, G.R., and Jolly, I.D. (1998). Impact of flooding on the water use of semi-arid riparian eucalypts. *Journal of Hydrology* **206**, 104-117.
- Amoros, C. and Bornette, G. (2002). Connectivity and biocomplexity in waterbodies of riverine floodplains. *Freshwater Biology* **47**, 761-76.
- Austin, A.T.; Yahdjian, L., Stark, J.M., Belnap, J., Porporato, A., Norton, U., Ravetta, D.A., and Schaeffer, S.M. (2004). Water pulses and biogeochemical cycles in arid and semiarid ecosystems. *Oecologia* **141**(2), 221-235.
- Bailey, P., Boon, P., and Morris, K. (2002). Australian Biodiversity-Salt Sensitivity Database. Land and Water Australia, Available at URL: <http://www.rivers.gov.au/research/contaminants/saltsen.htm>
- Bailey, P.C.E. and James, K. (2000). Riverine and Wetland Salinity Impacts-Assessment of R&D Needs. LWRRDC Occasional paper 25/99. Land and Water Resources Research and Development Corporation, Canberra.
- Baird, A.J., Price, J.S., Roulet, N.T., and Heathwaite, A.L. (2004). Special Issue of Hydrological Processes Wetland Hydrology and Eco-Hydrology. *Hydrological Processes* **18**(2), 211-212.
- Baird, R. (2005). Water and salt dynamics in a semi-arid floodplain wetland: Hart Lagoon, Waikerie, South Australia. Honours Thesis, School of Chemistry, Physics and Earth Sciences, Flinders University, Adelaide.
- Baldwin, D.S., Nielsen, D.L., Bowen, P.M., and Williams, J. (2005). Recommended Methods for Monitoring Floodplains and Wetlands. Murray-Darling Basin Commission, Canberra.
- Banks, E.W. (2003). Interactions between a saline lagoon and a semi-confined aquifer on Clarkes floodplain, South Australia. Honours Thesis, School of Chemistry, Physics and Earth Sciences, Flinders University, Adelaide.
- Bauer, P. (2004). Flooding and salt transport in the Okavango Delta, Botswana: key issues for sustainable wetland management. Phd Thesis, Swiss Federal Institute of Technology, Zurich.
- Bauer, P., Gumbricht, T., and Kinzelbach, W. (2006). A regional coupled surface water/groundwater model of the Okavango Delta, Botswana. *Water Resources Research* **42**, Paper W04403.
- Bayley, P.B. (1991). The flood pulse advantage and the restoration of river-floodplain systems. *Regulated Rivers: Research and Management* **6**, 75-86
- Bradley, C and Gilvear, D.J. (2000). Saturated and unsaturated flow dynamics in a floodplain wetland. *Hydrological Processes* **14**(16-17), 2945-2958
- Brock, M.A. (1981). The ecology of halophytes in the south-east of South Australia. *Hydrobiologia* **81**, 23-32.
- Brock, M.A. (1985). Are Australian salt lake systems different? Evidence from the submerged aquatic plant communities. *Proceedings of the Ecological Society of Australia* **14**, 43-50.
- Brock, M.A. and Lane, J.A.K. (1983). The aquatic macrophyte flora of saline wetlands in Western Australia in relation to salinity and permanence. *Hydrobiologia* **105**, 63-76.
- Brock, M.A., Nielsen, D.L., and Crosslé, K. (2005). Changes in biotic communities developing from freshwater wetland sediments under experimental salinity and water regimes. *Freshwater Biology* **50**(8), 1376-1390.
- Brock, M.A., Nielsen, D.L., Shiel, R.J., Green, J.D., and Langley, J.D. (2003). Drought and aquatic

community resilience: the role of eggs and seeds in sediments of temporary wetlands. *Freshwater Biology* **48**, 1207-1218.

- Brock, M.A. and Shiel, R.J. (1983). The composition of aquatic communities in saline wetlands in Western Australia. *Hydrobiologia* **105**, 77-84.
- Bullock, A. and Acreman, M. (2003). The role of wetlands in the hydrological cycle. *Hydrology and Earth Systems* **7**, 358-89.
- Butler, B.E. (1956). Parna - an aeolian clay. *Australian Journal of Science* **18**, 145-151.
- Cant, B., James, K., and Ryan, T. (2003). Salt Impact Model for Strategic Framework and Information to Indicate Biodiversity Thresholds to Salinity. Department of Sustainability and Environment, Arthur Rylah Institute, Heidelberg.
- Casanova, M.T. and Brock, M.A. (2000). How do depth, duration and frequency of flooding influence the establishment of wetland plant communities? *Plant Ecology* **147**(2), 237-250.
- Chesson, P., Gebauer, R.L.E., Schwinning, S., Huntly, N., Wiegand, K., Ernest, M.S.K., Sher, A., Novoplansky, A., and Weltzin, J.F. (2004). Resource pulses, species interactions, and diversity maintenance in arid and semi-arid environments. *Oecologia* **141**(2), 236-253.
- Clunie, P., Ryan, T., James, K., and Cant, B. (2002). Implication for Rivers From Salinity Hazards: A Scoping Study. Report to Murray-Darling Basin Commission. State Government of Victoria, Department of Natural Resources and Environment.
- Cramer, V. and Hobbs, R.J. (2002). Ecological consequences of altered hydrological regimes in fragmented ecosystems in southern Australia: Impacts and possible management responses. *Austral Ecology* **27**, 546-564.
- Dall'O', M., Kluge, W., and Bartels, F. (2001). FEUWAnet: a multi-box water level and lateral exchange model for riparian wetlands. *Journal of Hydrology* **250**, 40-62.
- Danielopol, D., Griebler, C., Gunatilaka, A., and Notemboom, J. (2003). Present state and future prospects for groundwater ecosystems. *Environmental Conservation* **30**(2), 104-130.
- Davis, J. A., Rosich, R. S., Bradley, J. S., Grown, J. E., Schmidt, L. G., and Cheal, F. (1993). Wetlands of the Swan Coastal Plain (Vol 6). Wetland Classification on the Basis of Water Quality and Invertebrate Community Data. Water Authority of Western Australia and WA Environmental Protection Authority, Perth.
- Davis, J.A. and Froend, R. (1999). Loss and degradation of wetlands in southern Australia: underlying causes, consequences and solutions. *Wetlands Ecology and Management* **7**, 13-23.
- Davis, J.A., McGuire, M., Halse, S.A., Hamilton, D., Horwitz, P., McComb, A.J., Froend, R.H., Lyons, M., and Sim, L. (2003). What happens when you add salt: Predicting impacts of secondary salinisation on shallow aquatic ecosystems by using an alternative-states model. *Australian Journal of Botany* **51**, 715-24.
- Dekker, S.C., Barendregt, A., Bootsma, M.C., and Schot, P.P. (2005). Modelling Hydrological Management for the Restoration of Acidified Floating Fens. *Hydrological Processes* **19**(20), 3973-3984.
- Doble, R., Simmons, C., Jolly, I., and Walker, G. (2006). Spatial relationships between vegetation cover and irrigation-induced groundwater discharge on a semi-arid floodplain, Australia. *Journal of Hydrology* **32**, 975-97.
- Ferraro, T.J. and Burgin, S. (1993). Review of environmental factors influencing the decline of Australian frogs. *Transacts of the Royal Society of NSW*. Surrey Beatty & Sons, Chipping Norton.

- Finlayson, C.M. and D'Cruz, R. (2005.). Inland Water Systems. In, *Ecosystems and Human Well-Being Wetlands and Water Synthesis : a Report of the Millennium Ecosystem Assessment*. (Eds R. Hassan, R. Scholes, and N. Ash.) World Resources Institute, Washington DC.
- Froend, R.H., Halse, S.A., and Storey, A.W. (1997). Planning for the recovery of Lake Toolibin, Western Australia. *Wetlands Ecology and Management* **5**, 73-85.
- Froend, R.H., Hedde, E.M., Bell, D.T., and McComb, A.J. (1987). Effects of salinity and waterlogging on the vegetation of Lake Toolibin, Western Australia. *Australian Journal of Ecology* **12**, 281-98.
- Froend, R.H. and McComb, A.J. (1991). An account of the decline of Lake Towerrinning, a wheatbelt wetland. *Journal of the Royal Society of Western Australia* **74**, 123-28.
- Froend, R.H. and van der Moezel, P.G. (1994). The impact of prolonged flooding on the vegetation of Coomalbidgup Swamp, Western Australia. *Journal of the Royal Society of Western Australia* **77**, 15-22.
- Garcia, A. (1999). Charophyte flora of south-eastern South Australia and south-western Victoria, Australia: systematics, distribution and ecology. *Australian Journal of Botany* **47**, 407-426.
- George, A. K., Walker, K. F., and Lewis, M. M. (2005). Population status of eucalypt trees on the River Murray floodplain, South Australia. *River Research and Applications* **21**, 271-282.
- George, R., Bennett, D., Wallace, K., and Cochrane, D. (1996). Hydrologic systems to manage salinity at the Toolibin and Towerrinning Lakes. 4th National Conference and Workshop on the Productive Use and Rehabilitation of Saline Lands, Albany, Promaco Conventions Pty Ltd: Perth.
- Ghassemi, F., Jakeman, A.J., and Nix, H.A. (1995). *Salinisation of land and water resources: human causes, extent, management and case studies*. University of New South Wales Press in association with Centre for Resource and Environmental Studies, Sydney.
- Gilvear, D.J., Andrews, R., Tellam, J.H., Lloyd, J.W., and Lerner, D.N. (1993). Quantification of the water balance and hydrogeological processes in the vicinity of a small groundwater fed wetland, East Anglia, UK. *Journal of Hydrology* **144**, 311-34.
- Haitjema, H.M. (1995). *Analytic Element Modeling of Groundwater Flow*. Academic Press, San Diego.
- Halse, S.A., Pearson, G.B., and Patrick, S. (1993a). Vegetation of Depth-Gauged Wetlands in Nature Reserves of Southwest Western Australia. Technical Report. Department of Conservation and Land Management, Perth.
- Halse, S.A., Ruprecht, J.K., and Pinder, A.M. (2003). Salinisation and prospects for biodiversity in rivers and wetlands of south-west Western Australia. *Australian Journal of Botany* **51**(6), 673-688.
- Halse, S.A., Williams, M.R., Jaensch, R.P., and Lane, J.A.K. (1993b). Wetland characteristics and waterbird use of wetlands in south-western Australia. *Wildlife Research* **20**, 103-126.
- Hamilton, S.K., Bunn, S.E., Thoms, M.C. and Marshall, J.C. (2005). Persistence of aquatic refugia between flow pulses in a dryland river system (Cooper Creek, Australia). *Limnology and Oceanography* **50**, 743-54.
- Hannah, D.M., Wood, P.J., and Sadler, J.P. (2004). Ecohydrology and hydroecology: A new paradigm? *Hydrological Processes* **18**(17), 3439-3445.
- Hart, B.T., Bailey, P., Edwards, R., Hortle, K., Kames, K., McMahon, A., Meredith, C., and Swadling, K. (1991). A review of the salt sensitivity of the Australian freshwater biota. *Hydrobiologia* **210**, 105-44.

- Hart, B.T., Lake, P.S., Webb, J.A., and Grace, M.R. (2003). Ecological risk to aquatic systems from salinity increases. *Australian Journal of Botany* **51**, 689-702.
- Hatton, T. and Evans, R. (1997). Dependence of Ecosystems on Groundwater and Its Significance to Australia. Occasional Paper No. 12/98. Land and Water Research and Development Corporation, Canberra.
- Hayashi, M. and Rosenberry, D.O. (2002). Effects of groundwater exchange on the hydrology and ecology of surface water (review paper). *Ground Water* **40**, 309-17.
- Hayashi, M., van der Kamp, G., and Rudolph, D.L. (1998a). Water and solute transfer between a prairie wetland and adjacent upland, 1. Water balance. *Journal of Hydrology* **207**, 42-55.
- Hayashi, M., van der Kamp, G., and Rudolph, D.L. (1998b). Water and solute transfer between a prairie wetland and adjacent uplands, 2. Chloride cycle. *Journal of Hydrology* **207**, 56-67.
- Herczeg, A.L., Dogramaci, S.S., and Leaney, F.W.J. (2001). Origin of dissolved salts in a large semi-arid groundwater system: Murray Basin, Australia. *Marine and Freshwater Research* **52**, 41-52.
- Heth, D. and MacRae, S. (1993). Selection of salt-tolerant clones of *Eucalyptus camaldulensis* Dehn. *South African Forestry Journal* **16**, 421-26.
- Holland, K.L., Tyerman, S.D., Mensforth, L.J., and Walker, G.R. (2006). Tree water sources over shallow, saline groundwater in the lower River Murray, south-eastern Australia: implications for groundwater recharge mechanisms. *Australian Journal of Botany* **54**, 193-205.
- Hunt, R.J., Bullen, T.D., Krabbenhoft, D.P., and Kendall, C. (1998). Using stable isotopes of water and strontium to investigate the hydrology of a natural and a constructed wetland. *Ground Water* **36**, 434-44.
- Hunt, R.J., Haitjema, H.M., Krohelski, J.T., and Feinstein, D.T. (2003). Simulating ground water-lake interactions: approaches and insights. *Ground Water* **41**, 227-37.
- Hunt, R.J., Krabbenhoft, D.P., and Anderson, M.P. (1996). Groundwater inflow measurements in wetland systems. *Water Resources Research* **32**, 495-507.
- Hunt, R.J., Walker, J.F., and Krabbenhoft, D.P. (1999). Characterizing hydrology and the importance of ground-water discharge in natural and constructed wetlands. *Wetlands* **19**, 458-72.
- Huxman, T.E., Snyder, K.A., Tissue, D., Leffler, A.J., Ogle, K., Pockman, W.T., Sandquist, D.R., Potts, D. L., and Schwinning, S. (2004). Precipitation pulses and carbon fluxes in semiarid and arid ecosystems. *Oecologia* **141**(2), 254-268.
- James, K.R., Cant, B., and Ryan, T. (2003). Responses of freshwater biota to rising salinity levels and implications for saline water management: a review. *Australian Journal of Botany* **51**(6), 703-713.
- James, K.R. and Hart, B.T. (1993). Effect of salinity on four freshwater macrophytes. *Australian Journal of Marine and Freshwater Research* **44**, 769-77.
- Jolly, I.D. and Walker, G.R. (1996). Is the field water use of *Eucalyptus largiflorens* F. Muell. affected by short-term flooding? *Australian Journal of Ecology* **21**, 173-183.
- Jolly, I.D. (1996). The effects of river management on the hydrology and hydroecology of arid and semi-arid floodplains. In, *Floodplain Processes*. (Eds M.G. Anderson, D.E. Walling, and P.D. Bates.) pp. 577-609. John Wiley & Sons Ltd., Chichester, New York.
- Jolly, I.D., McEwan, K.L., Cox, J., Walker, G.R., and Holland, K.L. (2002). Managing Groundwater and Surface Water for Native Terrestrial Vegetation Health in Saline Areas. CSIRO Land and

- Jolly, I.D., Narayan, K.A., Armstrong, D., and Walker, G.R. (1998). The impact of flooding on modelling salt transport processes to streams. *Environmental Modelling and Software* **13**, 87-104.
- Jolly, I.D., Walker, G.R., and Narayan, K.A. (1994). Floodwater recharge processes in the Chowilla Anabranch system. *Australian Journal of Soil Research* **32**, 417-35.
- Jolly, I.D., Walker, G.R., and Thorburn, P.J. (1993). Salt accumulation in semi-arid floodplain soils with implications for forest health. *Journal of Hydrology* **150**, 589-614.
- Joris, I. and Feyen, J. (2003). Modelling water flow and seasonal soil moisture dynamics in an alluvial groundwater-fed wetland. *Hydrology and Earth System Sciences* **7**(1), 57-66.
- Junk, W.J., Bayley, P.B., and Sparks, R.E. (1989). The flood pulse concept in river-floodplain systems. Proceedings of the International Large River Symposium, 106 pp. 110-127. Canadian Special Publications in Fisheries and Aquatic Science.
- Kefford, B. J., Papas, P. J., Metzeling, L., and Nuggeoda, D. (2004). Do Laboratory Salinity Tolerances of Freshwater Animals Correspond With Their Field Salinity? *Environmental Pollution* **129**(3), 355-362.
- Kingsford, R. T. (1997). Wetlands of the world's arid zones [A contribution from the Convention on Wetlands (Ramasar, Iram, 1971) to the First Session of the Conference of the Parties to the UN Convention to Combat Desertification .Briefing Paper].
- Krasnostein, A.L. and Oldham, C.E. (2004). Predicting wetland water storage. *Water Resources Research* **40**(10), Paper W10203.
- Lamontagne, S., Leaney, F.W. and Herczeg, A.L. (2005). Groundwater-surface water interactions in a large semi-arid floodplain: Implications for salinity management. *Hydrological Processes* **19**, 3063-80.
- Langevin, C., Swain, E., and Wolfert, M. (2005). Simulation of Integrated Surface-Water/Ground-Water Flow and Salinity for a Coastal Wetland and Adjacent Estuary. *Journal of Hydrology* **314** (1-4), 212-234.
- Marimuthu, S., Reynolds, D.A., and Le Gal La Salle, C. (2005). A field study of hydraulic, geochemical and stable isotope relationships in a coastal wetland system. *Journal of Hydrology* **315**, 93-116.
- McDonald, M.G. and Harbough, A.W. (2003). The history of MODFLOW. *Groundwater* **41**, 280-283.
- Mensforth, L.J. (1996). *Water use strategy of Melaleuca halmaturorum in a saline swamp*. PhD Thesis, The University of Adelaide.
- Mensforth, L.J., Thorburn, P.J., Tyerman, S.D., and Walker, G.R. (1994). Sources of water used by riparian *Eucalyptus camaldulensis* overlying highly saline groundwater. *Oecologia* **100**, 21-28.
- Mensforth, L.J. and Walker, G.R. (1996). Root dynamics of *Melaleuca halmaturorum* in response to fluctuating saline groundwater. *Plant and Soil* **184**, 75-84.
- Morris, J., Bickford, R., and Collopy, J. (1994). Tree and Shrub Performance and Soil Conditions in a Plantation Irrigated With Saline Groundwater. Department of Conservation and Natural Resources, Kew.
- Morris, J. D. (1984). Establishment of trees and shrubs on a saline site using drip irrigation. *Australian Forestry* **47**, 210-217.

- Mudd, G. M. (2000). Mound springs of the Great Artesian Basin in South Australia: A case study from Olympic Dam. *Environmental Geology* **39**(5), 463-476.
- Nield, S.P., Townley, L.R., and Barr, A.D. (1994). A framework for quantitative analysis of surface water-groundwater interaction: Flow geometry in a vertical section. *Water Resources Research* **30**, 2461-75.
- Nielsen, D.L., Brock, M.A., Rees, G.N., and Baldwin, D.S. (2003a). Effects of increasing salinity on freshwater ecosystems in Australia. *Australian Journal of Botany* **51**, 655-65.
- Nielsen, D. L., Brock. M. A., Crossle, K., Harris, K., Healey, M., and Jarosinski, I. (2003b). Does salinity influence aquatic plant and zooplankton communities developing from the sediment of two wetlands. *Freshwater Biology* **48**, 2214-2223.
- Niknam, S.R. and McComb, J. (2000). Salt tolerance screening of selected Australian woody species - a review. *Forest Ecology and Management* **139**, 1-19.
- Overton, I. C., Jolly, I. D., Slavich, P. G., Lewis, M. M., and Walker, G. R. (2006). Modelling vegetation health from the interaction of saline groundwater and flooding on the Chowilla floodplain, South Australia. *Australian Journal of Botany* **54**, 207-220.
- Pinder, A. M., Halse, S. A., McRae, J. M., and Shiel, R. J. (2005). Occurrence of aquatic invertebrates of the wheatbelt region of Western Australia in relation to salinity. *Hydrobiologia* **543**(1),1-24.
- Puckridge, J. T., Sheldon, F., Walker, K. F., and Boulton, A. J. (1998). Flow variability and the ecology of large rivers. *Marine and Freshwater Research* **49**, 55-72.
- Raisin, G., Bartley, J., and Croome, R. (1999). Groundwater influence on the water balance and nutrient budget of a small natural wetland in North Eastern Victoria, Australia. *Ecological Engineering* **12**, 133-47.
- Reid, M. A. and Brooks, J. J. (2000). Detecting Effects of Environmental Water Allocations in Wetlands of the Murray-Darling Basin, Australia. *Regulated Rivers: Research & Management* **16**(5), 479-496.
- Restrepo, J.I., Montoya, A.M., and Obeysekera, J. (1998). A wetland simulation module for the MODFLOW ground water model. *Ground Water* **36**(5), 764-770.
- Reynolds, J.F., Kemp, P.R., Ogle, K., and Fernández, R. J. (2004). Modifying the 'pulse-reserve' paradigm for deserts of North America: precipitation pulses, soil water, and plant responses. *Oecologia* **14**, 1194-210.
- Roberts, J., Young, B., and Marston, F. (2000). Estimating the Water Requirements for Plants of Floodplain Wetlands:a Guide. Occasional Paper 04/00. Land and Water Resources Research and Development Council, Canberra.
- Roberts, J. and Marston, F. (2000). Water Regime of Wetland and Floodplain Plants in the Murray-Darling Basin: A Source Book of Ecological Knowledge. Technical report 30/00. CSIRO Land and Water, Canberra.
- Rosenberry, D.O. and Winter, T.C. (1997). Dynamics of water-table fluctuation in an upland between two prairie potholes wetlands in North Dakota. *Journal of Hydrology* **191**, 266-89.
- Rutherford, J.C. and Kefford, B.J. (2005). Effects of Salinity on Stream Ecosystems: Improving Models for Macroinvertebrates. CSIRO Land and Water Technical Report 22/05, Canberra.
- Ryan, T. and Davies, P. (1996). Environmental Effects of Salinity and Nutrients From Salt Disposal: Approaches to the Development of Management Criteria. Flora and Fauna Technical Report No. 137, Department of Natural Resources and Environment , Melbourne.

- Sanders, A. (1991). Oral Histories Documenting Changes in Wheatbelt Wetlands. Occasional Paper 2/91. Department of Conservation and Land Management, Western Australia, Perth.
- Schwinning, S. and Sala, O.E. (2004). Hierarchy of responses to resource pulses in arid and semi-arid ecosystems. *Oecologia* **141**(2), 211-220.
- Schwinning, S., Sala, O.E., Loik, M.E., and Ehleringer, J.R. (2004). Thresholds, memory, and seasonality: Understanding pulse dynamics in arid/semi-arid ecosystems. *Oecologia* **141**(2), 191-193.
- Šimůnek, J., Šejna, M., and van Genuchten, M. (1999). The HYDRUS-2D Software Package for Simulating Water Flow and Solute Transport in Two-Dimensional Variably Saturated Media (Version 2.0). U.S. Salinity Laboratory, Riverside, California.
- Skinner, R., Sheldon, F. and Walker, K.F. (2001). Propagules in dry wetland sediments as indicators of ecological health: Effects of salinity. *Regulated Rivers: Research & Management* **17**(2), 191-197.
- Slavich, P.G. (1997). Modelling groundwater use, salt accumulation and vegetation growth in areas with shallow saline watertables. PhD Thesis, Flinders University of South Australia, Adelaide.
- Slavich, P.G., Walker, G.R., Jolly, I.D., Hatton, T.J., and Dawes, W.R. (1999a). Dynamics of *Eucalyptus largiflorens* growth and water use in response to modified watertable and flooding regimes on a saline floodplain. *Agricultural Water Management* **39**, 245-264.
- Slavich, P.G., Walker, G.R., and Jolly, I.D. (1999b). A flood history weighted index of average root-zone salinity for assessing flood impacts on health of vegetation on a saline floodplain. *Agricultural Water Management* **39**, 135-151.
- Smith, A.J. and Townley, L.R. (2002). Influence of regional setting on the interaction between shallow lakes and aquifers. *Water Resources Research* **38**, 1171.
- Sophocleous, M. (2002). Interactions between groundwater and surface water: the state of the science. *Hydrogeology Journal* **10**(1), 52-67.
- Strehlow, K., Davis, J., Sim, L., Chambers, J., Halse, S., Hamilton, D., Horwitz, P., McComb, A., and Freund, R. (2005). Temporal changes between ecological regimes in a range of primary and secondary salinised wetlands. *Hydrobiologia* **552**(1), 17-31.
- Taylor, P.J., Walker, G.R., Hodgson, G., Hatton, T.J., and Correll, R.L. (1996). Testing of a GIS Model of *Eucalyptus largiflorens* health on a semi-arid, saline floodplain. *Environmental Management*. **20**, 553-564.
- Thomson, L.A.J. (1988). *Salt tolerance in Eucalyptus camaldulensis and related species*. PhD Thesis, University of Melbourne.
- Thorburn, P. J., Hatton, T. J., and Walker, G. R. (1993). Combining measurements of transpiration and stable isotopes of water to determine groundwater discharge from forests. *Journal of Hydrology* **150**, 563-587.
- Thorburn, P.J. and Walker, G.R. (1994a). Variations in stream water uptake by *Eucalyptus camaldulensis* with differing access to stream water. *Oecologia* **100**, 293-301.
- Thorburn, P.J. and Walker, G.R. (1994b). Reliance of creek-side river red gums on creek water. *Australian Journal of Marine and Freshwater Research* **45**, 1439-1443.
- Thorburn, P.J., Walker, G.R., and Jolly, I.D. (1995). Uptake of saline groundwater by plants: an analytical model for semi-arid and arid areas. *Plant and Soil* **175**, 1-11.
- Thorburn, P.J. and Walker, G.R. (1993). The source of water transpired by *Eucalyptus camaldulensis*:

- Soil, groundwater or streams? In *Stable Isotopes and Plant Carbon - Water Relations*. (Eds J. R. Ehleringer, A. E. Hall, and G. D. Farquhar.) pp. 511-27. Academic Press, San Diego.
- Timms, B.V. and Boulton, A.J. (2001). Typology of arid-zone floodplain wetlands of the Paroo River (inland Australia) and the influence of water regime, turbidity, and salinity on their aquatic invertebrate assemblages. *Archiv Fur Hydrobiologie* **153**, 1-17.
- Tockner, K., Malard, F., and Ward, J.V. (2000). An extension of the flood pulse concept. *Hydrological Processes*. **14**(16-17), 2861-2883
- Toth, J. (1963). A theoretical analysis of groundwater flow in small drainage basins. *Journal of Geophysical Research* **68**: 4795-4812.
- Townley, L.R. and Davidson, M.R. (1988). Definition of a capture zone for shallow water table lakes. *Journal of Hydrology* **104**, 53-76.
- Townley, L.R. and Trefry, M.G. (2000) Surface water-groundwater interaction near shallow circular lakes: Flow geometry in three dimensions. *Water Resources Research* **36**(4), 935-948.
- Turner, J.V. and Townley, L.R. (2006). Determination of groundwater flow-through regimes of shallow lakes and wetlands from numerical analysis of stable isotope and chloride tracer distribution patterns. *Journal of Hydrology* **320**: 451-483.
- United Nations Environment Programme (1992). *World Atlas of Desertification*. UNEP, London.
- van Genuchten, M. (1978). Numerical Solutions of the One-Dimensional Saturated-Unsaturated Flow Equation. Research Report 78-WR-09. Water Resources Program, Department of Civil Engineering, Princeton University, Princeton, New Jersey.
- Walker, K.F. (1992). Chapter 22: The River Murray, Australia: A semiarid lowland river. In *The Rivers Handbook*. (Eds P. Calow and G. E. Petts.) pp. 472-92. Blackwell Scientific Publishers, Oxford..
- Walker, K.F., Sheldon, F., and Puckridge, J.T. (1995). A perspective on dryland river ecosystems. *Regulated Rivers: Research and Management* **11**, 85-104
- Walker, K.F. and Thoms, M.C. (1993). Environmental effects of flow regulation on the lower River Murray, Australia. *Regulated Rivers: Research and Management* **8**, 103-19.
- Walker, K.F., Thoms, M.C., and Sheldon, F. (1992). Effects of weirs on the littoral environment of the River Murray, South Australia. In *River Conservation and Management*. (Eds P. J. Boon, P. Calow, and G. E. Petts.) pp. 271-92. John Wiley & Sons, Australia.
- Wallace, K.J. (1997). *Case Study: Toolibin Lake and Catchment*. (Ed. K. J. Wallace.) Surry Beatty, Chipping Norton..
- Walton, R., Chapman, R.S., and Davis, J.E. (1996). Development and application of the wetlands dynamic water budget model. *Wetlands* **16**(3), 347-357.
- Whigham, P.A. and Young, W.J. (2001). Modelling river and floodplain interactions for ecological response. *Mathematical and Computer Modelling* **33**(6-7), 635-647.
- Williams, W.D. (1998a). Dryland wetlands. In *Wetlands for the Future*. (Eds A.J. McComb and J.A. Davis) pp. 33-47. Gleneagles Publishing, Adelaide.
- Williams, W.D. (1998b). Diversity and evolution of the fauna of dryland wetlands. In *Wetlands for the Future*. (Eds A.J. McComb and J.A. Davis) pp. 167-172. Gleneagles Publishing, Adelaide.
- Williams, W.D. (1999). Conservation of wetlands in drylands: A key global issue. *Aquatic Conservation: Marine and Freshwater Ecosystems* **9**, 517-22.

- Williams, W.D., Boulton, A.J., and Taaffe, R.G. (1990). Salinity as a determinant of salt lake fauna: a question of scale. *Hydrobiologia* **197**, 257-66.
- Winter, T.C. (1976). Numerical Simulation Analysis of the Interaction of Lakes and Ground Water. Professional Paper 1001. United States Geological Survey.
- Winter, T.C. (1978). Numerical simulation of steady-state, three dimensional ground-water flow near lakes. *Water Resources Research* **14**, 245-54.
- Winter, T.C. (1983). The interaction of lakes with variably saturated porous media. *Water Resources Research* **19**, 1203-18.
- Winter, T.C. (1999). Relation of streams, lakes, and wetlands to groundwater flow systems. *Hydrogeology Journal* **7**, 28-45.
- Winter, T.C. (2001). Ground Water and Surface Water: the Linkage Tightens, but Challenges Remain. *Hydrological Processes* **15**, 3605-6.
- Winter, T.C, Harvey, J.W., Franke, O.L, and Alley, W.M. (1999). *Ground water and surface water: a single resource*. U.S Geological Survey Circular 1139, Denver.
- Wurster, F.C., Cooper, D.J., and Sanford, W.E. (2003). Stream/aquifer interactions at Great Saad Dunes National Monument, Colorado: influences on interdunal wetland disappearance. *Journal of Hydrology* **271**, 77-100.
- Zimmermann, S., Bauer, P., Held, R., Kinzelbach, W., and Walther, J.H. (2006). Salt transport on islands in the Okavango Delta: Numerical investigations. *Advances in Water Resources* **29**, 11-29.