Hydrobiologia (2007) 591:147–164
DOI 10.1007/s10750-007-0805-4

SALT LAKES

Monitoring wetlands in a salinizing landscape: case studies from the Wheatbelt region of Western Australia

M. N. Lyons · S. A. Halse · N. Gibson · D. J. Cale · J. A. K. Lane · C. D. Walker · D. A. Mickle · R. H. Froend

Abstract Three elements of wetland biodiversity (aquatic invertebrates, waterbirds and overstorey vegetation of the wetland edge) have been monitored since 1998 at Lake Eganu and Paperbark Swamp in the Western Australian Wheatbelt to provide information about the changes occurring in wetland biodiversity in a landscape that is severely affected by dryland salinization. Changes in extent of wetland vegetation since the 1960s were examined using historical aerial photographs and waterbird use of Lake Eganu during the early 1980s was compared with recent waterbird survey results. Lake Eganu, which is within a major drainage line, started to become salinized in the mid-1960s, about 70 years after land clearing began in the catchment, and its salinity has increased an order of magnitude. The extent of wetland overstorey vegetation and the richness of freshwater aquatic invertebrates have both declined about 80%. Waterbird richness has also declined over the past 20 years, with changes in species composition. Salinization has not occurred at Paperbark Swamp, which is in a small catchment off the main drainage line, and there has been no consistent change in the biodiversity elements monitored.

Guest editors: John Tibby, Peter Gell, Lynda Radke, and Michael Reid
Salt Lakes: Salinity, Climate Change and Salinisation

M. N. Lyons · S. A. Halse (✉) · N. Gibson · D. J. Cale · D. A. Mickle
Science Division, Department of Environment and Conservation, PO Box 51, Wanneroo, WA 6946, Australia
e-mail: stuart.halse@bennelongia.com.au

J. A. K. Lane
Science Division, Department of Environment and Conservation, 14 Queen Street, Busselton, WA 6280, Australia

C. D. Walker
GEO & HYDRO Environmental Management, Suite 1, 9 Wygonda Road, Roleystone, WA 6111, Australia

R. H. Froend
Centre for Ecosystem Management, Edith Cowan University, Joondalup, WA 6027, Australia

Keywords Biodiversity · Vegetation health · Aquatic invertebrates · Waterbirds · Groundwater · Salinization · Landscape position

Introduction

The Wheatbelt of south-west Western Australia is a region of ca. 200,000 km^2 with a Mediterranean climate, an annual rainfall of 300–600 mm and an inland boundary corresponding to the limit of extensive land clearing (Fig. 1). It is widely recognized as a biodiversity hotspot because of rich floral diversity, much of which is under threat (Myers et al., 2000), and because of the diversity

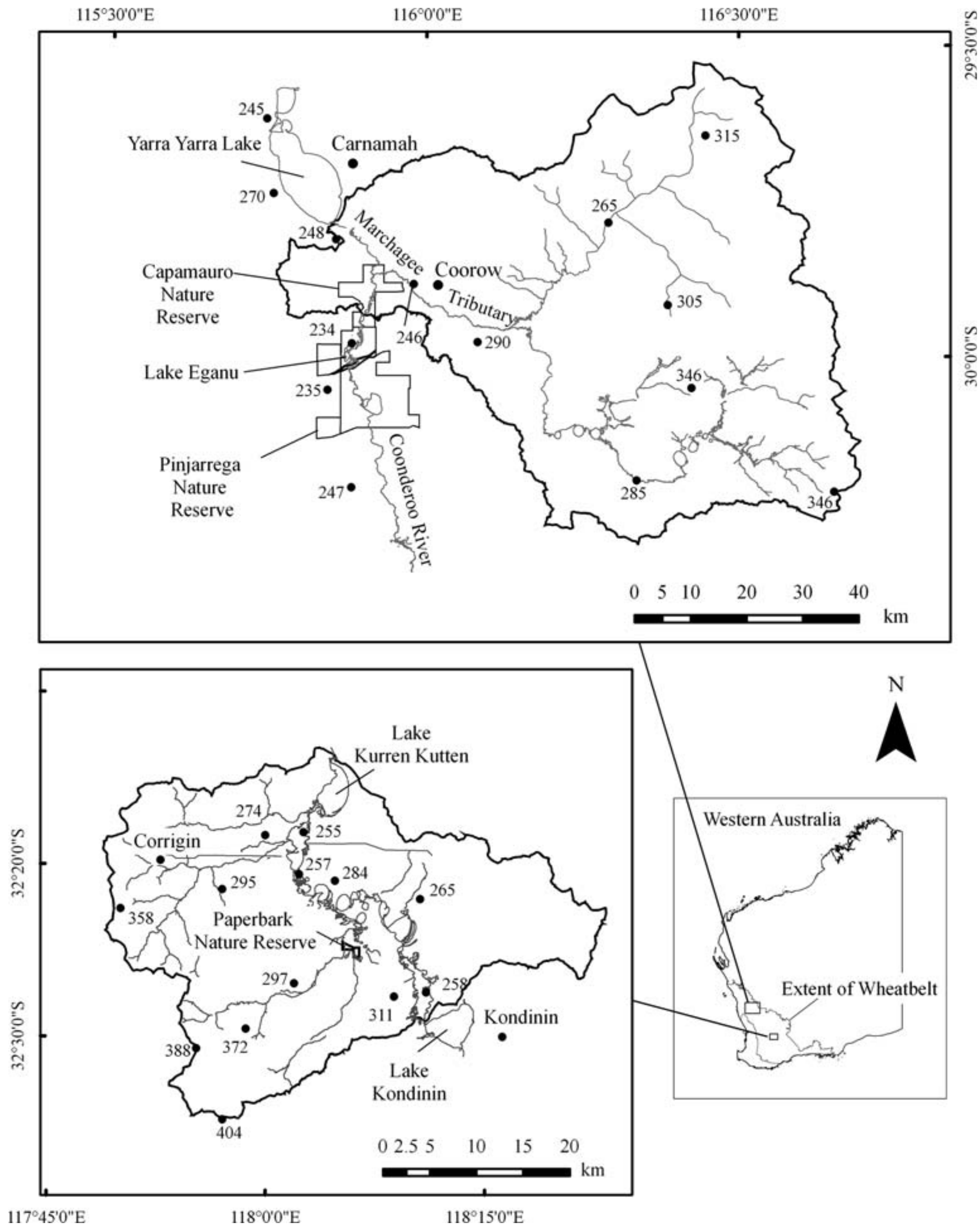

Fig. 1 Lake Eganu and Paperbark Swamp and their catchment settings. At Paperbark Swamp the small creek entering the reserve from the south flows into the wetland. Numbers indicate spot heights in metres

of many aquatic invertebrate groups, including Anostraca and Rotifera (Thomsen, 1999; Segers & Shiel, 2003). Prior to land clearing, vegetation consisted mostly of shrublands near the coast, open eucalypt woodlands across most of the landscape, and mallee in the south-eastern Wheatbelt (Beard, 1990; Gibson et al., 2004a). About 75% of this native vegetation has been removed during the past 120 years to enable growing of cereal crops and improved sheep pasture. Land clearing has produced substantial economic benefit, with Western Australia now being the biggest grain-growing state in Australia, sometimes producing more than 50% of Australia's total wheat yield (Anon, 2003). However, as in many other arid and semi-arid regions of the world, there has been an environmental cost in the form of dryland salinization (Abrol et al., 1988; Williams, 1999).

The mechanisms of salinization in the Wheatbelt are well documented (Teakle & Burvill, 1938; Ruprecht & Schofield, 1991; Clarke et al., 2002) and can be summarized as (1) native woodlands and shrublands are cleared and replaced with annual crops and pastures, (2) reduced annual evapo-transpiration leads to increased recharge to groundwater, (3) groundwater levels rise, salts stored within the regolith are mobilised, (4) spatial extent of saline groundwater discharge at the surface increases very markedly, and (5) significant changes occur in surface hydrology. Currently ca. 1 M ha of the Wheatbelt is salinized, increasing annually by ca. 14,000 ha (McFarlane et al., 2004). However, the distribution of salinization across the landscape is uneven, with wetlands, riparian zones, and the floors of broad valleys much more frequently affected than upland areas.

As first pointed out by Williams (1987), salinization presents a major and intractable threat to the biodiversity of both terrestrial and aquatic systems (see also George et al., 1995; Briggs & Taws, 2003; Cramer & Hobbs, 2002; McKenzie et al., 2003). Hart et al. (1991) comprehensively reviewed adverse effects of increased salinity on freshwater plants and animals, based mostly on laboratory studies. Subsequent field investigations in Australia have further demonstrated that salinization causes death of many riparian and lake-bed trees and shrubs, leading to more open riparian zones with altered plant species composition (Bell & Froend, 1990; Froend & McComb, 1991; Halse et al., 1993a; Lymbery et al., 2003; Lyons et al., 2004). Other field studies have shown salinization reduces the number and composition of aquatic macrophyte and other algal species within a wetland (Garcia, 1999; Blinn & Bailey, 2001; Blinn et al., 2004; Nielsen et al., 2003, 2004), richness and composition of aquatic invertebrates (Metzling et al., 1995; Skinner et al., 2001; Pinder et al., 2004, 2005), and richness and composition of waterbirds (Halse et al., 1993b, 2003). For riparian and emergent lake-bed plants, the effects of salinization may sometimes be mediated primarily through waterlogging and longer periods of inundation, rather than increased salinity (Bell & Froend, 1990; Froend & van der Moezel, 1994), while for waterbirds the loss of emergent and riparian vegetation may be the driver of change (Halse et al., 1993b, 2003).

A shortcoming of most existing evaluations of the impacts of salinization is that they were based on comparisons of salinized and unsalinized sites, rather than longitudinal studies of wetlands as they became salinized. Part of the Western Australian Government's response to salinization in the Wheatbelt was to initiate a long-term program to monitor elements of biodiversity and water quality at 25 wetlands (Government of Western Australia, 1996; Cale et al., 2004). Wetlands selected for monitoring had high conservation value (based on at least one aspect of biodiversity), were spread across the Wheatbelt, represented a range of wetland types and salinity conditions, and mostly had nearly 20 years record of depth and salinity (Lane & Munro, 1982). Data are being regularly collected on depth to groundwater, wetland depth and salinity, waterbird and aquatic invertebrate species composition, and overstorey vegetation health and species composition with the aims of (1) measuring trends in wetland condition, faunal composition, and vegetation health and composition at the 25 wetlands over time, as a representative sample of biodiversity trends at all Wheatbelt wetlands, and (2) relating trends at individual wetlands to anthropogenic activity, particularly degree of salinization, surrounding land-use and management actions at the wetland and in its catchment.

This paper presents initial results of monitoring at two wetlands, Lake Eganu and Paperbark Swamp, which have contrasting histories of salinization. We show that the degree of salinization at a wetland is more strongly related to position in the catchment than adjacent land-use. We also show that temporal scales of impacts differ among biodiversity elements, and that it may be decades before they are fully expressed.

Increasingly conservation decisions and management actions are being devolved to community-based organisations. Continued funding of the projects they are undertaking typically requires monitoring the effect of management actions on biodiversity values. In many cases, resources are limited and wetlands may be managed primarily for a single component of biodiversity. The current study, which integrates measurement of several environmental parameters and biodiversity elements, may assist community groups select appropriate elements of the biota to monitor. It also highlights the importance of understanding, before designing a monitoring program, the different kinds of responses to salinization that various biodiversity elements display.

Study areas

Lake Eganu (30°59′56″ E, 115°52′30″ S) is situated 420 km north of Perth within Pinjarrega Nature Reserve (4,686 ha), with Capamauro Nature Reserve (4,710 ha) adjoining to the north (Fig. 1). The lake occupies 110 ha, with a maximum depth of ca. 2.5 m, and is located on the Coonderoo River—a series of interconnected wetlands that consecutively fill and overflow—between Yarra Yarra Lake and Lake Pinjarrega (Fig. 1). It probably has only recently become seasonally connected to the regional watertable (Commander, 1981; Stelfox, 2004). Inflow from Yarra Yarra Lake into the Coonderoo River is estimated to occur only at intervals of about 50 years (Yesertener et al., 2000) and the most recent events are likely to have occurred in 1917 and 1918 (P. Muirden, unpublished data). The Marchagee tributary (Fig. 1), draining a catchment of 600,000 ha, of which 87% is cleared, accounts for 70% of inflow into Lake Eganu (mean 5.5 M m^3 y^{-1}, P. Muirden, unpublished data). Base stream flow salinities in the Marchagee tributary are 35–50 g l^{-1}. Land clearing began in the 1890s and was completed by the early 1960s (A. Doley, personal communication), except for some clearing peripheral to the nature reserve in the 1970s and 1980s. The topography of the catchment is highly subdued with elevations near the eastern watershed of the catchment of ca. 340 m falling to ca. 220 m on the lake margin (Fig. 1).

Paperbark Swamp (32°25′54″ S, 118°05′55″ E) is located 225 km south-east of Perth within Paperbark Swamp Nature Reserve (Fig. 1). The reserve covers 121 ha with the wetland, which has a maximum depth of 1.5 m, occupying approximately half this area. A small dam in the centre of the wetland extends 0.5 m below the lake-bed. The swamp is situated near a major paleo-drainage line between Kondinin Lake and Lake Kurrenkutten (Fig. 1). Although the wetland is at a similar elevation to the main paleo-drainage line (both ca. 255 m), it is perched above the regional watertable (Fig. 1). While surface water hydrology has not been studied, topographic interpretation suggests the wetland receives intermittent surface water flows via the poorly defined drainage line entering from elevated land (ca. 400 m) to the south-west (Fig. 1). Based on the salinity of the wetland, stream base-flows are assumed to be fresh (<2 g l^{-1}). Most of the catchment was cleared by the 1930s.

Methods

Inflow into wetlands in south-west Western Australia usually begins around July; wetlands reach maximal depths in September or October and then begin drying. Most years Lake Eganu retains water all year but Paperbark Swamp usually dries by the middle of summer (i.e., January). For convenience, we refer to monitoring any time during the hydrological cycle by the year of inflow, e.g. July 1997 to February 1998, as 1997.

Wetland depth and salinity

Monitoring of wetland depth and salinity commenced in September 1979 (Eganu) and November 1982 (Paperbark). Measurements were made in September and November to capture depth maxima, with salinity measured as Total Dissolved Solids using a Hamon salinity bridge, calibrated against seawater. After 1994, instruments used were a Type MC5 Ale salinity-temperature bridge and TPS LC-81 and TPS 90-FLMV conductivity meters with commercially prepared salinity standards. All measurements were standardized to 25°C. Rainfall data were compiled for the nearest two available meteorological stations to each wetland (Bureau of Meteorology, 2004).

Analysis was focused on detecting changes in wetland depth and salinity over time. Because of the very large increase in salinity at low depths in Lake Eganu, caused by evapo-concentration (see Halse, 1981), salinity values at this wetland were compared only between years when depth was ≥2 m. At Paperbark Swamp inspection of the data showed little evapo-concentration effect, and only years when the wetland was dry were excluded from analysis. Salinity data were transformed [log (salinity +0.0001)] to achieve a normal distribution of residuals and a regression of salinity against time (number of months since first measurement) was calculated for September and November values and pooled data.

Trends in wetland depth were examined by comparing depth data with mean rainfall data of the two nearest weather stations to each wetland. These were Carnamah (39 k N) and Coorow (20 k NE) townsites for Lake Eganu and Corrigin (46 k NW) and Kondinin (36 k SE) townsites for Paperbark Swamp.

Shallow groundwater monitoring

Shallow groundwater monitoring bores were established adjacent to vegetation transects (see below). Two bores were installed beside each transect approximately 10–15 m towards the transect centre from the upslope and downslope ends. Bores were made of 45 mm diameter PVC pipe with the lower section slotted and the hole sealed with blue metal and bentonite clay. Depth to groundwater was measured at least twice yearly to capture seasonal maxima and minima—usually about March and September. Depth to groundwater was measured using an electrical conductivity dipper tape and salinity (as electrical conductivity, mS $^{-1}$) and pH were measured using a TPS-W81 meter. Groundwater monitoring commenced in December 1999 at Lake Eganu and March 2000 at Paperbark Swamp.

Vegetation

Available historical aerial photography was examined to document the past condition of vegetation at the wetlands, with consecutive images overlain to examine changes in vegetation over time. The oldest available photography was from 1959 for Lake Eganu and 1962 for Paperbark Swamp. There was additional photography from 1969, 1974, 1981, 1996 and 2001 for Lake Eganu and 1972, 1983, 1996 and 2001 for Paperbark Swamp.

At both wetlands three radial transects, positioned to sample representative stands of trees and shrubs in the wetland basin and riparian zone, were sampled in 1998, 2001 and 2004. Elevation profiles were determined every 4 m along both sides and the centre of transects using a dumpy level and staff, with the data transformed to height above the lowest point of the wetland. At Lake Eganu, transects 1 and 2 sampled the riparian zone (Fig. 2) while transect 3 was located in a small wetland ca. 3 km upstream. At Paperbark Swamp, transects 1 and 2 sampled the riparian zone and transect 3 sampled the wooded wetland lake-bed.

The transects themselves consisted of two or three contiguous 20 × 20 m plots, each subdivided into five 4 × 20 m subplots. Within transects, all trees and large understorey shrubs were permanently marked with a uniquely numbered tag attached by nail or wire. Diameters of the five largest stems of each individual tree or shrub were measured at the tag (usually breast height, 1.37 m) and the plant's basal area was calculated as the

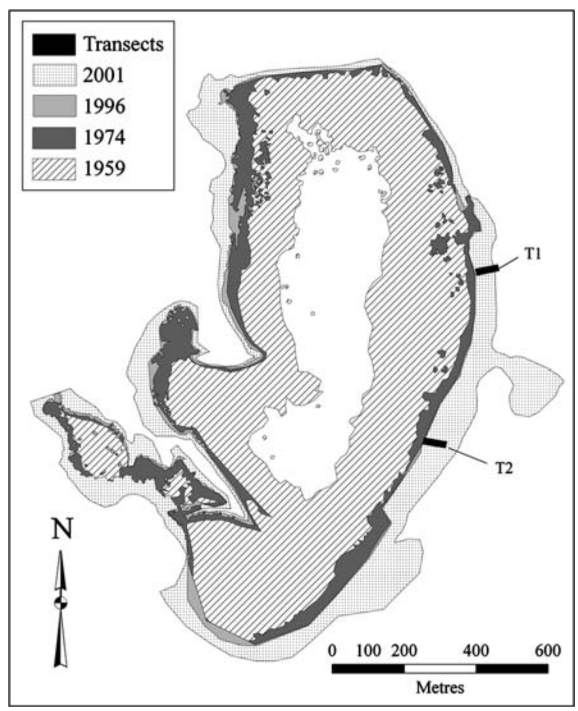

Fig. 2 Sequential decline in tree cover within the basin of Lake Eganu. The outer boundary is the limit of riparian vegetation, based on the approximate upslope limit of flooding as determined from aerial photographs

sum of the five cross sectional areas. Diameter at breast height (DBH) was not measured for stems with DBH <0.02 m. For each plant, an assessment of crown condition was made using a visual scoring system based on the scheme of Grimes (1978). This scheme has been used widely in Australia with various modifications (see Eldridge et al., 1993; Stone et al., 2003). Three of the original five components (crown density, dead branches and epicormic growth) were scored in the current study and summed to give an aggregate condition score. Our scheme closely follows that of Stone et al. (2003) except that both crown density and dead branches were scored on a five point scale with values of 9, 7, 5, 3, 1. Epicormic growth was scored on a five point scale with values of 5-1, with an additional category 'epicormic growth severe on crown and stem' (score 1). The final composite score for a healthy/vigorous tree was 23 and a score of 3 represented an individual close to death. The number of seedlings/saplings (largest stem <0.02 m DBH) of overstorey taxa were counted within each subplot.

Changes in basal area along vegetation transects were examined by plotting total basal area (all species in a stand) for each sample year and investigating the relationships between basal area, transect position and elevation. Differences in tree or shrub health between sampling years were assessed by one-way repeated measures analysis of variance using a Friedman non-parametric test (StatSoft, 2001). To track stand condition over the time-frame of the study, trees that died between sampling events were retained and given a condition rating of zero.

Aquatic invertebrates and waterbirds

Faunal monitoring occurred every second year (1998, 2000, 2002, 2004 at Lake Eganu; 1999, 2001, 2003 at Paperbark Swamp). Data collected in 1997 during a regional survey of the Wheatbelt (Pinder et al., 2004) were used to supplement monitoring data at Paperbark Swamp.

At each wetland during late winter, spring and autumn (provided water was present), one or two observers counted all waterbirds present using binoculars and spotting scopes. Observations at Lake Eganu were made from a small punt when water levels were high, otherwise observers walked around and through the wetland (Cale et al., 2004). Change in waterbird use over time at Lake Eganu was examined by comparing recent monitoring counts with data from the early 1980s (Jaensch et al., 1988). For each of the 12-month periods between July 1981 and June 1985, a late winter, spring and autumn count was selected to provide a dataset equivalent to recent monitoring data (both 3 seasons sampling for 4 years, i.e. 12 surveys) and the number of species and total number of waterbirds seen each year were calculated. The validity of comparing 1980s and recent monitoring is questionable, given that Robertson & Massenbauer (2005) showed recent counts at Lake Wheatfield in the southern Wheatbelt by Birds Australia, the organization that undertook the 1980s surveys (Jaensch et al., 1988), recorded only a third of the number of species per survey seen by us (Cale et al., 2004) during the same time period. We suggest that it is unlikely the counting discrepancies at Lake Eganu were as large because the wetland is more accessible and the Birds Australia observers were particularly experienced. However, comparing our recent counts with 1980s surveys by Jaensch et al. (1988) may have under-estimated the changes in waterbird richness that have occurred. Insufficient counting was done at Paperbark Swamp by Jaensch et al. (1988) to enable an analysis of changes in waterbird use there.

Invertebrates were sampled during spring, in the same years as the waterbird counts, at two widely spaced sites within each wetland (see Halse et al., 2002). At each site, a benthic sub-sample (250 μm mesh, D-frame pondnet—all habitats including sediment sampled) and a plankton sub-sample (50 μm mesh, D-frame pondnet—all habitats excluding sediment sampled) were collected. Each sub-sample consisted of 50 m of sweeping along a 200 m path. The benthic sub-sample was preserved in 70–80% ethanol and the plankton sub-sample was fixed in 5% borax-buffered formalin. In the laboratory, sub-samples were separated into three size fractions (benthic sample—2 mm, 250 μm and 500 μm, plankton sample—250 μm, 90 μm and 53 μm) and representative specimens of all species were picked out using a dissecting microscope with 10–50× magnification. Animals were identified to species or morphospecies using dissecting and compound microscopes, keys and voucher collections (see Cale et al., 2004). Change in invertebrate composition at Lake Eganu over time was examined by comparing the species list of Halse (1981) with the species composition of monitoring samples.

Datasets underlying the analyses presented below are available from the Librarian, Department of Environment and Conservation, Locked Bag 104, Bentley Delivery Centre, WA 6983, Australia.

Results

Wetland depth and salinity

Long term data from weather stations near Lake Eganu since 1912 show annual rainfall has been variable, with 3 years or periods of >600 mm (1917–1918, 1963 and 1999) and no long-term trends (e.g. 392 mm y^{-1} pre-1964 vs. 377 mm y^{-1} 1964 and later, t = 0.49, NS, 91 df). Since monitoring began, Lake Eganu has held water during most Septembers and Novembers (spring) and has often held water throughout the year. The lake was dry in September and November 1980 and November 2002 (Fig. 3a), while 1999 resulted in flows of about 30 M m^3 into and through the lake (Yesertener et al., 2000).

Based on early records from similar wetlands, Lake Eganu was likely to have been fresh when full and perhaps sub-saline as it dried until the 1960s, although no data on salinity exist. By November 1978, just before water quality monitoring began, salinity had risen to 10 g l^{-1} (Halse, 1981). Since 1981, water salinity has increased significantly (Fig. 4). Although excluded from the trend analysis, salinity of the wetland when depth was ca. 1.0 m ranged from 100 g l^{-1} to 150 g l^{-1}, and water often became super-saturated with salt when the wetland was near dry. There was no significant reduction of depth over time.

Rainfall during the period 1982–2004 in the vicinity of Paperbark Swamp, during which lake depth and salinity have been monitored, was similar to the period 1918–1981 (average 360 mm y^{-1} vs. 353 mm y^{-1}, t = 0.70, NS, 85 df). Paperbark Swamp was dry in September and November 1985, 1987, 1997, 1998, 2001 and 2002 (Fig. 3b). No significant trends in salinity or depth were observed.

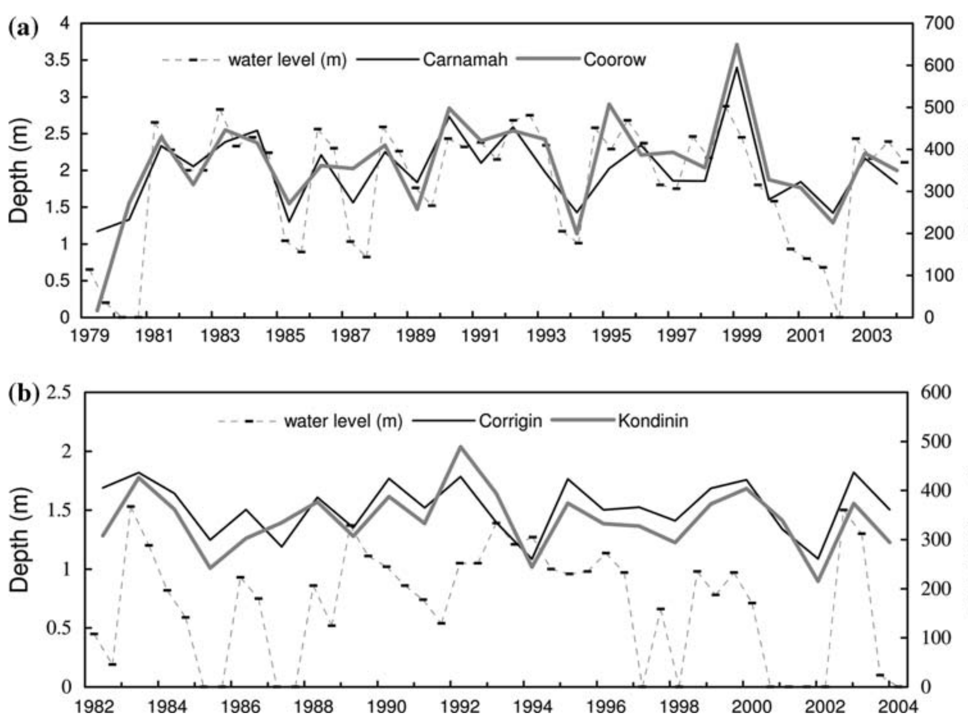

Fig. 3 Wetland depth measured in September and November each year plotted against total annual rainfall of the two nearest meteorological stations (see text). (**a**) Lake Eganu. (**b**) Paperbark Swamp

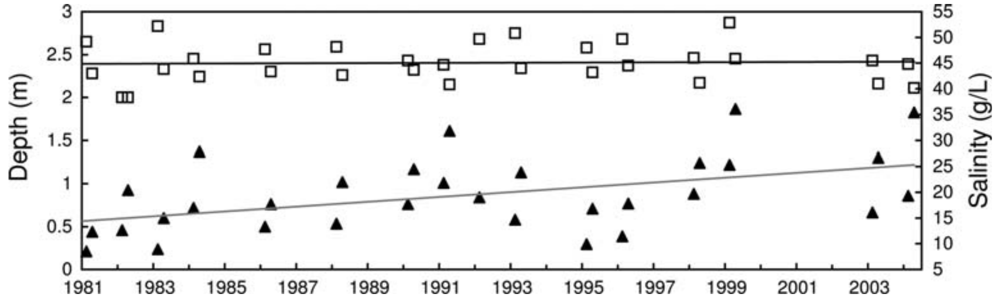

Fig. 4 Lake Eganu salinity (*solid triangles*) and depth (*open squares*) trends. Data for lake depths <2 m excluded. Linear regressions for salinity data were significant for September ($r = 0.53$, $P < 0.05$, 14 df), November ($r = 0.61$, $P < 0.05$, 13 df) and all data pooled ($r = 0.48$, $P < 0.01$, 27 df). Depth showed no significant change with time

Shallow groundwater

Groundwater in transects 1 and 2 at Lake Eganu was <1 m below ground level in 1999 but declined to almost 2 m below ground during 2000–2002, probably as a result of low rainfall (Fig. 3a and 5a). Groundwater rose close to the surface in spring 2004 and earlier differences in depth to groundwater between transects 1 and 2 and transect 3 disappeared. Although there was a general decline in groundwater salinity over the monitoring period, salinity showed a fairly consistent pattern of variation relative to rainfall (Fig. 3a) and was reduced by seasonal recharge (Fig. 5b). The maximum conductivity of 8,150 m S m^{-1} was recorded at transect 2 from groundwater within 0.5 m of the surface (Fig. 5b).

Depth to groundwater at Paperbark Swamp mostly varied between 2 m and 4 m at all bores (Fig. 6a). Groundwater depths reflected rainfall only weakly, perhaps because an impervious clay layer below the wetland prevents in situ recharge (Fig. 3b and 6a). Maximum groundwater conductivity of 7,700 m S m^{-1} was similar to Lake Eganu (Fig. 6b).

Vegetation

Lake Eganu

Prior to salinization, Lake Eganu was a wooded wetland. The dominant trees *Casuarina obesa* Miq and *Melaleuca strobophylla* Barlow formed relatively open woodland but localized stands of recruits would have had almost closed canopies. Unsalinized examples of these wooded swamps and lakes have little other emergent vegetation within the wetland basin, and halophytes are uncommon. Herbaceous communities occur at the margins as wetlands dry over late spring and summer. Dense thickets of other *Melaleuca* species often occur about the upper flood level. Upslope of the maximum flood level, riparian vegetation gives way to terrestrial vegetation. Lunette dunes, supporting terrestrial vegetation, are common.

Comparison of the historical aerial photographs of Lake Eganu revealed dramatic loss of vegetation from the lake-bed and most upstream wetlands along the Coonderoo River. Transects 1 and 2 were situated at the present day limit of *C. obesa* occurrence with *Halosarcia* spp. extending further downslope into the wetland basin. In 1959 live trees extended 250 m farther onto the lake-bed than currently occurs and *C. obesa*, with some *M. trobophylla* and *Eucalyptus rudis* Endl at the upslope margin, dominated the lake-bed vegetation. The centre of the wetland has been open water since at least 1959 (Fig. 2). Photographs showed the greatest loss of tree cover occurred between 1959 and 1974 (1969 photography was of poor quality and difficult to interpret). By 2001, photographs showed total absence of live trees from the main basin of the wetland (Fig. 2). Live stands of *C. obesa* were restricted to a narrow zone (mostly <50 m wide) at the upper limit of wetland filling.

Detailed measurements of extant riparian vegetation at Lake Eganu showed significant change occurred during the monitoring period. The overstorey in transect 2 was a pure stand of *C. obesa* and

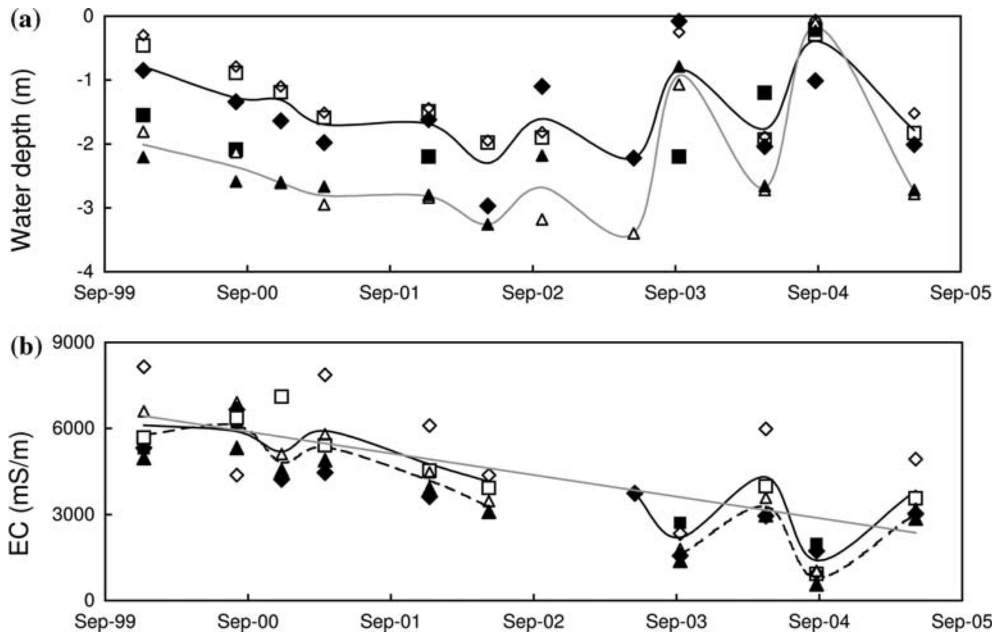

Fig. 5 Lake Eganu groundwater. (**a**) Average groundwater depth for the bores on transects 1 (*solid diamonds*, *solid squares*) and 2 (*open squares*, *open diamonds*) is shown as solid black line. Average groundwater depth for the bores on transect 3 (*open triangles*, *solid triangles*) is the lower grey line. (**b**) Average electrical conductivity for bores on transects 1 and 2 is the solid black line. Average electrical conductivity for bores on transect 3 is the broken line. The fitted linear regression line for data from bore 1 at transect 2 (*open squares*) is significant ($r = 0.81$, $P < 0.01$, 7 df). Regressions for other bores were very similar and not shown

its basal area over the entire transect declined by 78% between 1998 and 2004. When plotted against elevation profile, losses of basal area occurred at the lowest parts of the transect (Fig. 7). Transect 1 showed less decline in basal area but, where decline occurred, it was observed at lower parts of the transect within subplots 1E, 2A and 2B (Fig. 7). Transect 3, at a small unsalinized wetland upstream of Lake Eganu, accumulated basal area (except for death of a large tree due to lightning) over the monitoring period (data not shown).

Over the monitoring period, tree health showed significant decline at all Lake Eganu transects (Table 1). Mean health scores of *C. obesa* declined in transects 1 and 2. *Melaleuca strobophylla* at transect 3 showed a significant decline in health rating between 1998 and 2001 (Table 1). *Eucalyptus loxophleba* occurred at transect 1 but there were too few individuals to test for statistical significance. Very little germination and subsequent seedling recruitment was observed for overstorey species within transects or the main body of Lake Eganu.

Wetland depth data were combined with transect elevation profiles to examine the pattern of transect inundation since 1979. At Lake Eganu transect 2, which was closest to the water body, was completely inundated on three occasions (September 1983, September 1993 and September 1999). Partial inundation occurred during an additional 11 years (Fig. 8a). During partial inundation events wetland salinity ranged from 8.5 g l^{-1} to 36 g l^{-1}. Salinities recorded at times of total inundation were 8.9, 15 and 25 g l^{-1}. At transect 1, it is likely limited inundation occurred within the swale (between 22 m and 30 m along the transect) during September 1999 when wetland salinity was 25 g l^{-1}. Inundation history at transect 3 could not be compiled because there were no depth data for this subsidiary wetland.

Paperbark Swamp

At Paperbark Swamp, the extent and density of large canopies of *M. strobophylla* was readily discernible from aerial photography, and showed

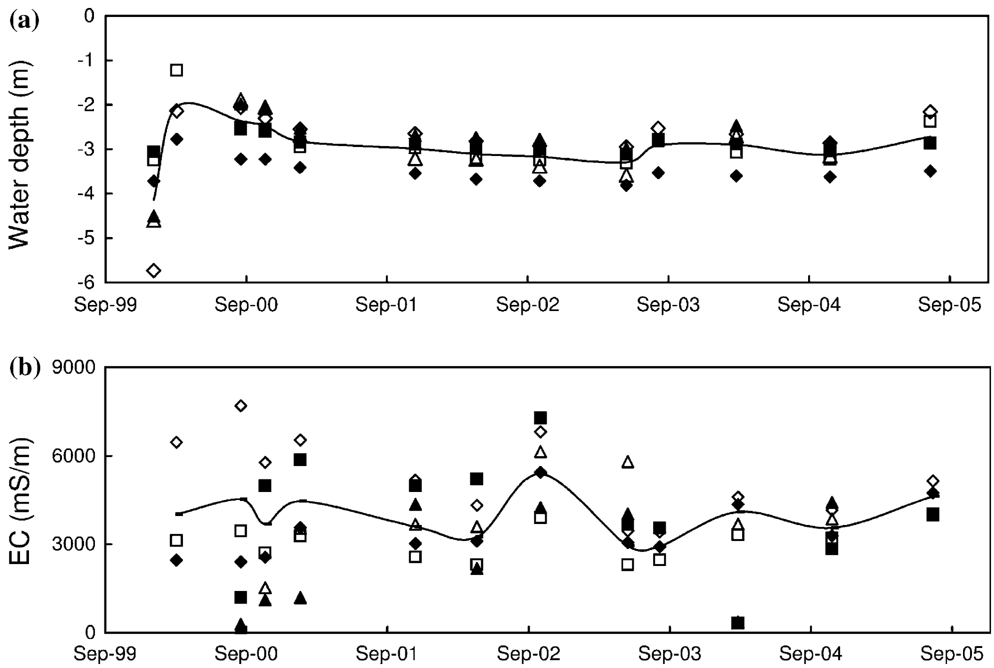

Fig. 6 Paperbark Swamp groundwater. (**a**) Average groundwater depth for the all bores over the three transects is shown as the solid black line (first set of measurements taken soon after bore installation and probably represent pre-equilibration values) (transect 1, *solid diamonds* and *solid squares*; transect 2, *open squares* and *open diamonds*; transect 3, *open triangles* and *solid triangles*). (**b**) Electrical conductivity of groundwater with average value for bore 1 on transect 1 and both bores on transect 2 shown as solid black line

little change in tree cover in the main body of the wetland from 1962 to 2001. Some decline of mature *M. strobophylla* had occurred at the southern end, near the inflow, and a stand of *Melaleuca phoidophylla* (Craven) appeared to have thinned out. Minor changes also occurred outside the reserve, including additional clearing in the north-eastern corner and some earthworks in the vicinity of transect 3 that reduced flooding from the wetland onto adjacent farmland. Here *M. strobophylla* expanded into a previously cleared area.

There was little change in basal area within transects at Paperbark Swamp between 1998 and 2004. At most transects basal area increased or remained stable. *Melaleuca strobophylla* on the lake-bed (transect 3) showed a significant change in health scores between sample times (Table 1), resulting from a small decline in vigour in 2001. When pooled across the wetland, health scores for *M. trobophylla* showed a significant difference between years (Table 1). Comparison of health scores over time for *E. loxophleba* (Benth) and *E. yilgarnensis* Brooker (pooled owing to low numbers of trees at individual transects) showed no significant change (Table 1). Large numbers of seedling *M. strobophylla* (273 plants, 0.01–0.3 m tall) were observed in transect 3, on the lake-bed, during 2004 sampling. This recruitment event is most likely to have occurred following inundation of transect 3 during 2003 (see below). In previous years only a single seedling was observed at transect 1.

Water depth data indicated transect 3 (on the lake-bed) was totally inundated during September and November 1983, 1989, 1992, 1993, 2003 and September 1990, 1994 and 1996 (Fig. 8b). Wetland salinity values during these events ranged from 0.1 g l^{-1} to 0.4 g l^{-1}. The maximum wetland depth, recorded in 1983, would have produced depths at transect 3 of between 0.5 m and 0.7 m.

Aquatic invertebrates

Aquatic invertebrate species composition between 1998 and 2004 at Lake Eganu was typical of a salinized wetland (see Pinder et al., 2004).

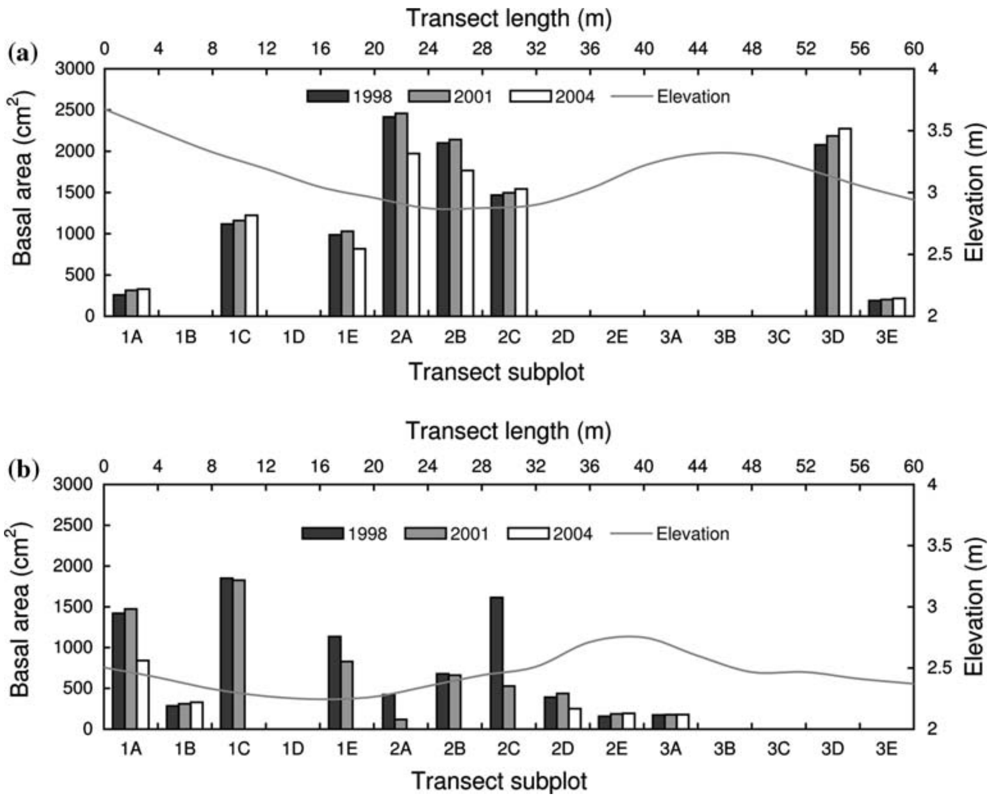

Fig. 7 Total basal area (all species combined) over time at Lake Eganu. (**a**) Transect 1. (**b**) Transect 2. Transect subplots (1A–3E) represent each 4 m section within transects. Elevation profiles for transects 1 and 2 are equivalent to wetland water depth

Table 1 χ-square values (N, df) for Friedman non-parametric ANOVA of tree health ratings over sampling times (1998/1999, 2001/2002 and 2004/2005)

Wetland	Transect 1	Transect 2	Transect 3	All
Paperbark Swamp				
Melaleuca strobophylla	2.35 (11,2)NS	–	11.57 (29,2)*	10.03 (40,2)**
Eucalyptus loxophleba	–	–	–	2.29 (9,2)NS
E. yilgarnensis	–	–	–	1.81 (7,2)NS
Eucalyptus spp. pooled	–	–	–	4.07 (16,2)NS
Lake Eganu				
Casuarina obesa	100.54 (96,2)‡	25.22 (28,2)‡	20.75 (32,2)***	63.60 (172,2)‡
Melaleuca strobophylla	–	–	17.33 (13,2)***	–

NS, $P > 0.05$; *, $P < 0.05$; **, $P < 0.001$; ***, $P < 0.0005$; ‡, $P < 0.0001$

Altogether, 39 species were recorded, with a negative relationship between invertebrate richness and salinity among monitoring events ($r = -0.85$, $P < 0.10$, 2 df). Unfortunately, the only previous invertebrate sampling of the lake occurred in 1979, with low sampling effort and during a year when the lake was probably atypically saline for that decade. All seven species collected in 1979 (see Halse 1981) were recorded again between 1998 and 2004, suggesting there has been no major shift in invertebrate composition beyond the reduction in richness expected because of increased salinity.

A total of 123 aquatic invertebrate species were recorded at Paperbark Swamp during the pre-monitoring 1997 survey by Pinder et al. (2004) (33

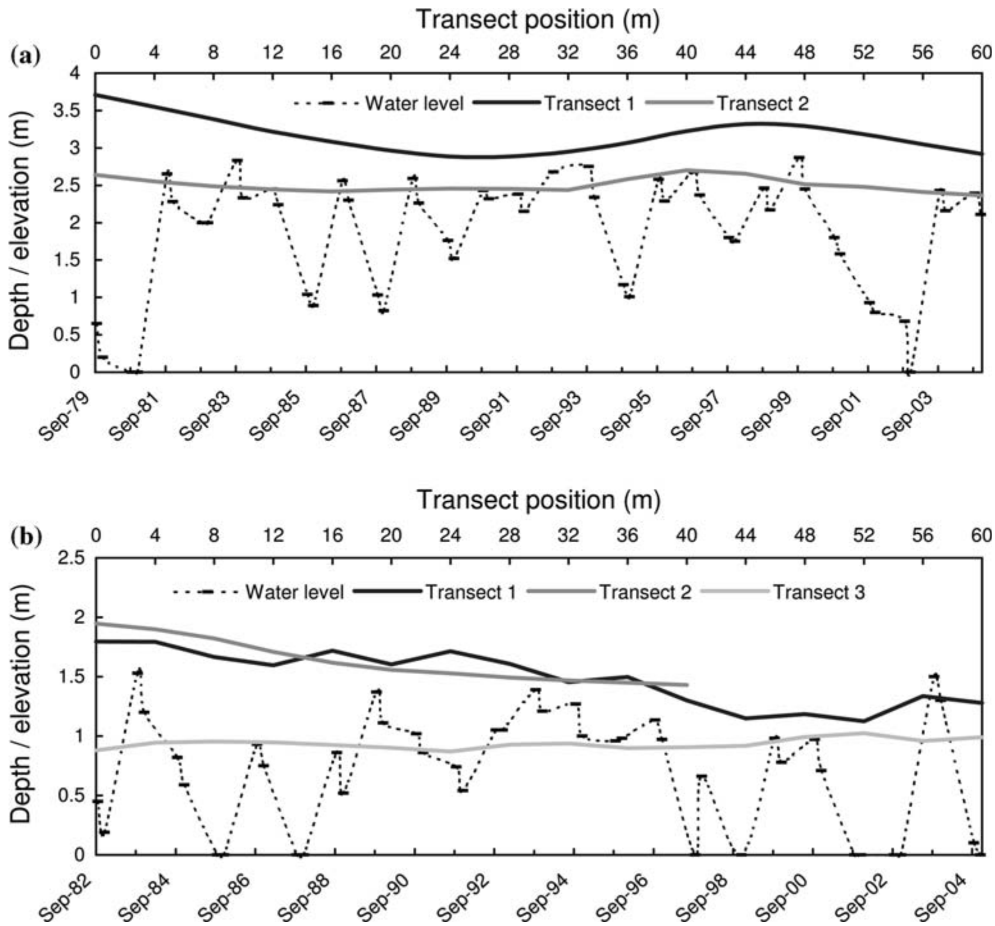

Fig. 8 Wetland surface water depth and inundation of transects. (**a**) Lake Eganu. (**b**) Paperbark Swamp. Elevation profiles of transects are overlain. By projecting each depth horizontally the extent of inundation of transects at a given time can be seen

species) and the 1999 and 2003 monitoring events (77 and 64 species, respectively). The difference in species richness between 1997 and the monitoring events is probably the result of only half the sampling effort being applied (Halse et al., 2002) and minimal habitat diversity because only the small excavated dam contained water in 1997 (depth was 0.5 m compared with 0.9 and 1.4 m during later sampling). There was little variation in wetland salinity across surveys.

Waterbirds

Twenty-seven waterbird species were recorded at Lake Eganu during the monitoring and Jaensch et al. (1988) surveys analysed herein, with a maximum count during one survey of 10,638 individuals. There was a small decline in species richness between 1981–1984 and 1998–2004 but no change in abundance (Fig. 9). The most likely explanation for the decline in species richness is the slow increase in salinity levels at Lake Eganu since 1979 (Fig. 4).

The decline in species richness was accompanied by the disappearance of nine fresh and hyposaline water species after 1984 (*Oxyura australis* Gould, *Chenonetta jubata* (Latham), *Ardea pacifica* Latham, *Erythrogonys cinctus* Gould, *Phalacrocorax melanoleucos* (Vieillot), *Platalea flavipes* Gould, *Stictonetta naevosa* (Gould), *Chlidonias hybridus* (Pallas), *Himantopus himantopus* (Linnaeus), see Halse et al., 1993b) and the arrival of three salt-water specialists (*Larus novaehollandiae* Stephens, *Charadrius*

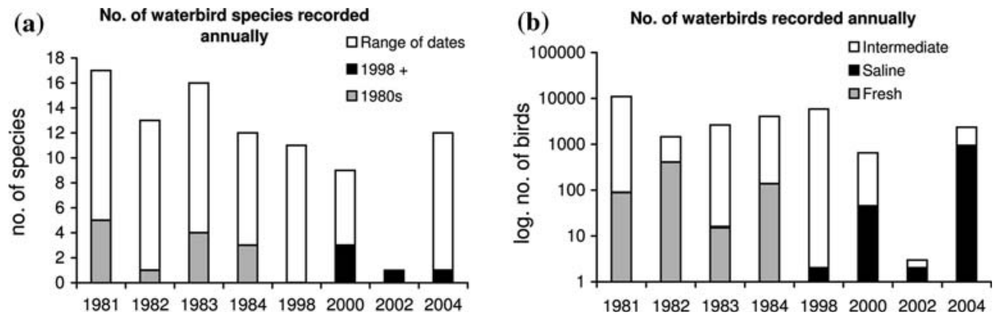

Fig. 9 Changes in waterbird use of Lake Eganu. (**a**) Number of species recorded annually, showing species recorded only during the 1980s, during 1998 or later, and during both periods. Richness was significantly higher in the 1980s ($t = 2.3$, $P < 0.05$, 6 df). (**b**) Abundance of individual waterbirds recorded annually, showing decline in abundance of fresh or hyposaline water species (see text) across time and increase in salt-adapted species

ruficapillus Temminck, *Cladorhynchus leucocephalus* (Vieillot)) (Fig. 9b). There was a strong negative relationship between species richness and salinity (log-transformed) across the individual surveys that made up the annual datasets ($r = -0.84$, $P < 0.001$, 19 df). This correlation was not an artefact of a depth—salinity relationship; results were similar when surveys at depths <1 m were excluded ($r = -0.77$, 17 df). The sole species attempting to breed at Lake Eganu between 1998 and 2004 was *Anas gracilis* Buller.

Ten waterbird species were recorded at Paperbark Swamp during the seven monitoring surveys when water was present. The maximum number of individual waterbirds counted in a single survey was 269. Nine species were recorded in both 1999 and 2003 (the 2 years when water was present for all three monitoring counts), which was similar richness to that of Lake Eganu in recent years (Fig. 9). Five species were recorded breeding (*Anas superciliosa* Gmelin, *A. gracilis*, *Malacorhynchus membranaceus* (Latham), *Fulica atra* Linnaeus, *A. pacifica*. Paperbark Swamp was surveyed once between 1981 and 1984, with *A. gracilis* being the only species recorded and there is no evidence of a decline in waterbird value in recent years.

Discussion

Salinization

The ecological consequences of dryland salinization of rivers and wetlands in the Wheatbelt are likely to be widespread (Halse et al., 2003), because most wetlands occur low in the landscape within broad valley floors and their associated drainage lines. Lake Eganu lies within a large, well-vegetated nature reserve but this natural vegetation has not protected the lake from salinization arising in agricultural parts of the catchment. Despite numerical modelling having shown clearly that hydrological balance in salinized catchments will be achieved only when recharge to groundwater across most of the catchment returns to pre-clearing values (Clarke et al., 2002; George et al., 2002), much management action to protect biodiversity from the effects of salinization is based on local actions, such as the protection of remnant vegetation and tree planting, in an attempt to control watertables near to biodiversity assets. Lake Eganu is a salutary reminder that, for many wetlands, these efforts will have limited capacity to achieve protection.

Studies of the mechanisms of salinization usually emphasize rise of groundwater (Wood, 1924; Mulcahy 1978) so that proximity of a saline water table to the lake-bed, with consequent waterlogging of vegetation and percolation of salt into wetlands, is often regarded as the main mechanism by which wetlands become salinised. However, most salinization at Lake Eganu has been the result of saline surface-water inflows. The lake overflows only after extreme rainfall events, so that in most years saline inflows incrementally increase salt load. Local discharge of salt from saline groundwater through the lake-bed has probably begun only recently (see below) but it has raised the severity of salinization to a new level.

Salinization proceeds in a step-wise fashion, with changes wrought by large rainfall episodes and associated flooding and increase in groundwater levels (Lewis, 1998). Most vegetation decline at Lake Eganu occurred in the 1960s (Fig. 2), probably as a result of high annual rainfall in 1963 (666 mm) causing a large salt load to be delivered to the wetland. A similar phenomenon occurred at Coomalbidgup Swamp on the south coast of Western Australia in the late 1980s, when rising groundwater in the catchment increased run-off and caused prolonged flooding in the swamp, with concomitant vegetation death (Froend & van der Moezel, 1994). Anecdotal accounts suggest that the pattern of salinization at Lake Eganu, with excessive surface water followed years later by elevated groundwater, is typical of many Wheatbelt wetlands (NARWC, 1987; Sanders, 1991), although detailed study is likely to reveal differences in the salinization process between wetlands that reflect local hydrology.

Unlike Lake Eganu, which is located in the main drainage line of a large catchment, Paperbark Swamp is situated within a small, elevated sub-catchment that is hydrologically isolated from the main drainage line. For this reason, despite being surrounded by cleared agricultural land and located in a reserve only twice the area of the wetland, it has shown no sign of salinization. Not all wetlands receiving inflows from elevated catchments are likely to be protected from salinization, however. Depending on local scale hydrogeology saline groundwater discharge can develop high in the landscape (Bettenay et al., 1964; Gibson et al., 2004b).

The importance of landscape position in structuring physicochemical and biological attributes of lakes has been emphasized in other unsalinized temperate catchments with similarly low relief to the Wheatbelt (Kratz et al., 1997; Webster et al., 2000). Lakes low in the landscape/regional flow pathway show higher concentrations of base cations because of greater contribution of groundwater relative to precipitation (Kratz et al., 1997). This process is strongly amplified in salinizing catchments because of naturally large regolith salt loads, highly saline groundwaters and the cumulative mobilisation of salt along regional flow pathways.

Regolith salt loads throughout the Wheatbelt are very high, ranging from 100 tonnes ha^{-1} near the coast to 1,000–10,000 tonnes ha^{-1} in the poorly drained, subdued catchments further inland (George et al., 1995). As alluded to in the previous paragraph, landscape distribution of regolith salt is very variable at smaller scales, although valley floors typically hold the largest storages. In very large catchments of the Wheatbelt, however, total regolith salt storage across the catchment is likely to be so large that local variations in regolith are likely to be irrelevant to salinization outcomes. This is the case at Lake Eganu. However, some small catchments may have particular regolith characteristics (e.g. low salt storage due to high elevation and a large depth to bedrock) that limit the development of secondary salinization (see McFarlane & George, 1992). Although we have no information on the regolith of the Paperbark Swamp catchment, regolith characteristics may have contributed to persistence as a freshwater wetland.

Management efforts to protect wetland biodiversity in salinized landscapes are likely to be more successful if hydrological setting of wetlands is taken into account when choosing wetlands intended as biodiversity refuges. The disadvantage is that avoiding wetlands associated with major drainage lines will lead to few large wetlands being protected, despite them usually being held in higher regard by the general community (see Lane et al., 1996) and being critical to maintaining overall size of waterbird populations (see Halse et al., 1995).

Biodiversity impacts

The responses of vegetation, aquatic invertebrates and waterbirds to salinization at Lake Eganu are incompletely documented but appear to show roughly parallel trends, despite the three biotic elements having different life histories. This universal response is the result of high salt levels being toxic to all freshwater, and associated emergent and riparian, organisms (see Hart et al., 1991; Lymbery et al., 2003). In contrast, the biotic elements show variable patterns of response to environmental parameters such as water depth, nutrients and colour that merely affect competitive gradients and predatory interactions (see Davis et al., 2001).

Overstorey tree and shrub species of lake-bed and riparian zone at Lake Eganu have contracted substantially in distribution but still remain around the wetland margin at high elevations, albeit in declining health (Fig. 2). This provides, through the skeletons of dead trees, an easily accessed record of the original community. Many understorey species, particularly annuals, are likely to show more rapid and complete responses to salinization. Unsalinized examples of *C. obesa* dominated wooded wetlands studied elsewhere in the wheatbelt contained 41 species (assemblages 2.1 and 3.1 of Lyons et al., 2004) that are susceptible to salinization and a proportion of these probably occurred at Lake Eganu prior to the 1960s and subsequently disappeared.

Most aquatic invertebrates, especially freshwater species, have relatively narrow salinity tolerances; consequently, species richness fluctuated during monitoring according to annual wetland salinity and undoubtedly has been reduced significantly since salinity began to increase at Lake Eganu. Based on the assumption that water salinity at Lake Eganu was usually about 1 g l^{-1} in October in the early 1960s, the predictive equation of Pinder et al. (2005) suggests salinization had reduced median richness almost 80% by the late 1990s. Much of this species loss would have occurred in the initial stages of salinization (see Pinder et al., 2005), but richness probably showed substantial variation from year-to-year as salinization progressed, reflecting rainfall-induced annual fluctuations in salinity. The apparent similarity of species composition during 1979, when all species collected belonged to the widespread, salt-tolerant Wheatbelt assemblages 13 and 6 identified by Halse et al. (2004), to that collected during later monitoring surveys suggests the fauna has been typical of salinized wetlands for the past 25 year and, although more affected by variations in annual conditions than overstorey vegetation, is exhibiting a similar kind of response to salinization.

The mobility of waterbirds has meant that their presence on a wetland is often regarded as being as much influenced by conditions elsewhere as by those in the wetland. However, waterbird species richness at Lake Eganu was well correlated with annual wetland salinity and community composition showed changes between the early 1980s and late 1990s that appeared to reflect a combination of the loss of habitat provided by overstorey vegetation and increasing salinity (Fig. 9; Halse et al., 1993b). It is likely that substantially more species would have used Lake Eganu prior to the 1960s and that its importance for breeding would have been high. Fresh lakes with live vegetation are the preferred breeding habitat of most waterbird species in the Wheatbelt (see Halse, 1987; Halse et al., 1993b), as was reflected by the much greater use of Paperbark Swamp for breeding than Lake Eganu despite more species and individuals occurring at the latter wetland.

While the Lake Eganu data demonstrate that waterbirds have value as indicators of overall biodiversity trends in salinized wetlands, there is some doubt whether they respond in the same way as other biodiversity elements in the early stages of salinization. Studies comparing many wetlands of different types and salinities in south-west Western Australia suggest that waterbird richness may increase with salinity up to ca. 10 g l^{-1}, after which richness declines (Halse et al., 1993b, 2003). This seems intuitively likely, particularly at closed sedge swamps, highly turbid claypans and darkly stained humic wetlands, where salinization brings about favourable habitat changes (opening of vegetation and increased visibility of food items) that may over-ride the detrimental physiological effects of higher salinities. It should be noted, however, that substitution of fresh water species by salt tolerant ones may mask waterbird responses if species richness alone is used as an indicator. Further time-series studies of the effects of salinization on waterbird communities are required. It should also be noted that substitution of fresh water species by salt tolerant species may mask changes where species richness alone is used as an indicator.

Vegetation and groundwater interactions

One of the objectives of monitoring was to document the effect of high groundwater levels on vegetation health. The short records available do not allow many inferences to be drawn but the vigour of vegetation at Paperbark Swamp suggests that saline groundwater needs to be within ca. 1 m of the surface before significant decline will occur (see Fig. 6). Vegetation condition at transect 3 at Lake Eganu suggests that short periods of high groundwater do not affect vegetation if salinities are low (Fig. 5). Measurements at Toolibin Lake, farther south in the Wheatbelt, provided similar results (Bell & Froend, 1990). While it appears very likely that the very high groundwater levels now experienced at Lake Eganu itself (Fig. 5; Stelfox, 2004) are causing further death of overstorey vegetation, more measurements (from several wetlands) are required to quantify the relationship, which will vary according to species, between groundwater salinity, micro-topography, soil type and perhaps rainfall (see Cramer & Hobbs, 2002).

Conclusion

The design of monitoring programs should be driven primarily by the questions being asked (Walters, 1986) and monitoring at Lake Eganu and Paperbark Swamp has included all major components of hydrology and aquatic biology (hardly any fish species occur in the Wheatbelt) because the objective was to document trends in aquatic biodiversity values across the Wheatbelt, largely in relation to hydrological change. However, many smaller monitoring programs intended to document the effects of local actions will have insufficient funding to measure more than one aspect of biodiversity.

Results from Lake Eganu suggest that all three biodiversity elements monitored there provide useful information and can be used to infer changes in each other. We suggest factors relevant to choosing a single element to monitor at a wetland are:

(1) the element should be relatively species-rich or spatially extensive at the wetland, so that trends can be reliably detected,
(2) if the wetland is being managed primarily for a particular biodiversity element, that element should be monitored,
(3) the sensitivity of the element to changes in wetland conditions should match the timescale over which changes in wetland condition are expected as a result of management activities. For example, if management is likely to improve conditions within a year, waterbirds or aquatic invertebrates would be more suitable elements for monitoring than vegetation, and
(4) monitoring is more effective if there is sufficient information to model the expected behaviour of the element being monitored prior to management action, so that monitoring results may be compared against a predicted outcome.

Acknowledgements A. Clarke, B. Franke, R. Gurner, G. Ogden, G. Pearson, M. Pennifold, K. Sutcliffe and Y. Winchcombe variously assisted with field work and data analysis. K. Wallace provided comments on an ealier draft and N. Huang assisted in preparing the figures. Funding was provided through the Western Australian State Salinity Strategy.

References

Abrol, I. P., J. S. P. Yadav & F. I. Massoud, 1988. Salt-affected soils and their management., FAO Soils Bulletin 39. Food and Agriculture Organization, United Nations, Rome.

Anon, 2003. Australian Yearbook 2003. Australian Bureau of Statistics, Canberra.

Beard, J. S., 1990. Plant Life of Western Australia. Kangaroo Press, Sydney.

Bell, D. T. & R. H. Froend, 1990. Mortality and growth of tree species under stress at Lake Toolibin in the Western Australian Wheatbelt. Journal of the Royal Society of Western Australia 72: 63–66.

Bettenay, E., A. V. Blackmore & F. J. Hingston, 1964. Aspects of the hydrological cycle and related salinity in the Belka Valley, Western Australia. Australian Journal of Soil Research 2: 187–210.

Blinn, D. W. & P. C. E. Bailey, 2001. Land-use influence on stream water quality and diatom communities in Victoria, Australia: A response to secondary salinization. Hydrobiologia 466: 231–244.

Blinn, D. W., S. A. Halse, A. M. Pinder, R. J. Shiel & J. M. McRae, 2004. Diatom and micro-invertebrate communities and environmental determinants in the Western Australia wheatbelt: a response to salinization. Hydrobiologia 528: 229–248.

Briggs, S. V. & N. Taws, 2003. Impacts of salinity on biodiversity—clear understanding or muddy confusion? Australian Journal of Botany 51: 609–617.

Bureau of Meteorology, 2004. Australian Monthly Rainfall. Bureau of Meteorology, Canberra (available on CD).

Cale, D. J., S. A. Halse & C. D. Walker, 2004. Wetland monitoring in the Wheatbelt of south-west Western

Australia: Site descriptions, waterbird, aquatic invertebrate and groundwater data. Conservation Science Western Australia 5: 20–135.

Clarke, C. J., R. J. George, R. W. Bell & T. J. Hatton, 2002. Dryland salinity in south-western Australia: Its origins, remedies, and future research directions. Australian Journal of Soil Research 40: 93–113.

Commander, P., 1981. The hydrogeology of the Eneabba area Western Australia. Unpublished MSc Thesis, University of Western Australia, Perth.

Cramer, V. A. & R. J. Hobbs, 2002. Ecological consequences of altered hydrological regimes in fragmented ecosystems in southern Australia: Impacts and possible management strategies. Australian Ecology 27: 546–564.

Davis, J. A., S. A. Halse & R. H. Froend, 2001. Factors influencing biodiversity in coastal plain wetlands of southwestern Australia. In Gopal, B., W. J. Junk & J. A. Davis (eds), Biodiversity in Wetlands: Assessment, Function and Conservation. Backhuys, Leiden, 89–100.

Eldridge, S. R., P. J. Thorburn, K. L. McEwan & T. J. Hatton, 1993. Health and structure of *Eucalyptus* communities on Chowilla and Monoman Isalnds of the River Murray Floodplain, South Australia. Divisional Report 93/3. CSIRO Divison of Water Resources, Canberra.

Froend, R. H. & A. J. McComb, 1991. An account of the decline of Lake Towerrinning, a wheatbelt wetland. Journal of the Royal Society of Western Australia 73: 123–128.

Froend, R. H. & P. G. van der Moezel, 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, R. J., D. J. McFarlane & R. J. Speed, 1995. The consequences of a changing hydrologic environment for native vegetation in southwestern Australia. In Saunders, D. A., J. L. Craig & E. M. Mattiske (eds), Nature Conservation 4: The Role of Networks. Surrey Beatty & Sons, Sydney, 9–22.

George, R. J., C. J. Clarke & T. J. Hatton, 2002. Computer modelled groundwater response to recharge management for dryland salinity control in Western Australia. Advances in Environmental Monitoring and Modelling 2: 3–35.

Gibson, N., G. J. Keighery, M. N. Lyons & A. Webb, 2004a. The terrestrial flora and vegetation of the Western Australian wheatbelt. Records of the Western Australian Museum Supplement 67: 139–202.

Gibson, N., G. J. Keighery & J. A. K. Lane, 2004b. Five years of monitoring of the Lake Muir-Unicup wetland system, south-western Australia. Journal of the Royal Society of Western Australia, 87: 29–33.

Government of Western Australia, 1996. Western Australian Salinity Action Plan. Government of Western Australia, Perth.

Grimes, R. J., 1978. Crown assessment of natural spotted gum (*Eucalyptus maculata*) ironbark (*E. fibrosa, E. drepanophylla*) forest. Technical Paper No. 7. Queensland Forest Service, Brisbane.

Halse, S. A., 1981. Faunal assemblages of some saline lakes near Marchagee, Western Australia. Australian Journal of Marine and Freshwater Research 32: 133–142.

Halse, S. A., 1987. Probable effect of increased salinity on the waterbirds of Lake Toolibin. Technical Report 15. Department of Conservation and Land Management, Perth.

Halse, S. A., G. B. Pearson & S. Patrick, 1993a. Vegetation of depth-gauged wetlands in nature reserves of south-west Western Australia. Technical Report 30. Department of Conservation and Land Management, Perth.

Halse, S. A., M. R. Williams, R. P. Jaensch & J. A. K. Lane, 1993b. Wetland characteristics and waterbird use of wetlands in south-western Australia. Wildlife Research 20: 103–126.

Halse, S. A., G. B. Pearson, R. M. Vervest & F. H. Yung, 1995. Annual waterfowl counts in south-west Western Australia—1991/92. CALMScience 2: 1–24.

Halse, S. A., D. J. Cale, E. J. Jasinska & R. J. Shiel, 2002. Monitoring change in aquatic invertebrate biodiversity: sample size, faunal elements and analytical methods. Aquatic Ecology 36: 395–410.

Halse, S. A., J. K. Ruprecht & A. M. Pinder, 2003. Salinization and prospects for biodiversity in rivers and wetlands of south-west Western Australia. Australian Journal of Botany 51: 673–688.

Halse, S. A., M. N. Lyons, A. M. Pinder & R. J. Shiel, 2004. Biodiversity patterns and their conservation in wetlands of the Western Australian Wheatbelt. Records of the Western Australian Museum Supplement 67: 337–364.

Hart, B. T., P. Bailey, R. Edwards, K. Hortle, K. James, A. McMahon, C. Meredith & K. Swadling, 1991. A review of the salt sensitivity of the Australian freshwater biota. Hydrobiologia 210: 105–144.

Jaensch, R. P., R. M. Vervest & M. J. Hewish, 1988. Waterbirds in nature reserves of south-western Australia 1981–1985: Reserve accounts. Report 30. Royal Australasian Ornithologists Union, Melbourne.

Kratz, T. K., K. E. Webster, C J Bowser, J. J. Magnuson & B. J. Benson, 1997. The influence of landscape position on lakes in northern Wisconsin. Freshwater Biology 37: 209–217.

Lane, J. A. K. & D. R. Munro, 1982. 1981 review of rainfall and wetlands in the south-west of Western Australia. Report 56. Department of Fisheries and Wildlife, Perth.

Lane J., R. Jaensch & R. Lynch, 1996. Western Australia. In Blackley R., S. Usback & K. Langford (eds), A Directory of Important Wetlands in Australia, 2nd edn. Australian Nature Conservation Agency, Canberra, 759–843.

Lewis, F., 1998. Modelling direct episodic recharge in the Western Australian Wheatbelt. Resource Management Technical Report 168. Department of Agriculture, Perth.

Lymbery, A. J., R. G. Doupe & N. E. Pettit, 2003. Effects of salinization on riparian plant communities in experimental catchments on the Collie River, Western Australia. Australian Journal of Botany 51: 667–672.

Lyons, M. N., N. Gibson, G. J. Keighery & S. D. Lyons, 2004. Wetland flora and vegetation of the Western Australian Wheatbelt. Records of the Western Australian Museum Supplement 67: 39–90.

McFarlane, D. J. & R. J. George, 1992. Factors affecting dryland salinity in two wheatbelt catchments in Western Australia. Australian Journal of Soil Research 30: 85–100.

McFarlane, D. J., R. J. George & P. A. Caccetta, 2004. The extent and potential area of salt-affected land in Western Australia estimated using remote sensing, digital terrain models. In Dogramaci, S. & A. Waterhouse (eds), Engineering Salinity Solutions, 1st National Salinity Engineering Conference 2004. Conference Proceedings. Insitute of Engineers, Australia, 55–60.

McKenzie, N. L., A. H. Burbidge & J. K. Rolfe, 2003. Effect of salinity on small, ground-dwelling animals in the Western Australian wheatbelt. Australian Journal of Botany 51: 725–740.

Metzling, L., T. Doeg & W. O'Connor, 1995. The impact of salinization and sedimentation on aquatic biota. In Bradstock, R. A., T. D. Auld, R. T. Kingsford, D. Lunney & D. P. Siversten (eds), Conserving Biodiversity: Threats and Solutions. Surrey Beatty & Sons, Sydney, 126–136.

Mulcahy, M. J., 1978. Salinization in the southwest of Western Australia. Search 9: 269–272.

Myers, N., R. A. Mittermeier, C. G. Mittermeier, G. A. B. da Foncesa & J. Kents, 2000. Biodiversity hotspots for conservation priorities. Nature 403: 853–858.

NARWC, 1987. The Status and Future of Lake Toolibin as a Wildlife Reserve. Water Authority of Western Australia, Perth.

Nielsen, D. L., M. A. Brock, G. N. Rees & D. S. Baldwin, 2003. Effects of increasing salinity on freshwater ecosystems in Australia. Australian Journal of Botany 51: 655–665.

Nielson, D. L., M. A. Brock, K. Crossle, K. Harris, M. Healey & J. Jarosinski, 2004. The effects of salinity on aquatic plant germination and zooplankton hatching from two wetland sediments. Freshwater Biology 48: 2214–2223.

Pinder, A. M., S. A. Halse, J. M. McRae, & R. J. Shiel, 2004. Aquatic invertebrate assemblages of wetlands and rivers in the wheatbelt region of Western Australia. Records of the Western Australian Museum Supplement 67: 7–37.

Pinder, A. M, S. A. Halse, J. M. McRae & R. J. Shiel, 2005. Occurrence of aquatic invertebrates of the wheatbelt region of Western Australia in relation to salinity. Hydrobiologia 543: 1–24.

Robertson, D. & T. Massenbauer, 2005. Applying hydrological thresholds to wetland management for waterbirds, using bathymetric surveys and GIS. In Zerger, A. & R. M. Argent (eds), MODSIM 2005 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand, December 2005, pp. 2407–2413. ISBN: 0-9758400-2-9. http://www.mssanz.org.au/modsim05/papers/robertson_2.pdf.

Ruprecht, J. R. & N. J. Schofield, 1991. Effects of partial deforestation on hydrology and salinity in high salt storage landscapes. I. Extensive block clearing. Journal of Hydrology 129: 19–38.

Sanders, A., 1991. Oral histories documenting changes in Wheatbelt wetlands. Occasional Paper 2/91. Department of Conservation and Land Management, Perth.

Segers, H. & R. J. Shiel, 2003. Microfaunal diversity in a biodiversity hotspot: New rotifers from Western Australia. Zoological Studies 42: 516–521.

Skinner, R., F. Sheldon & K. F. Walker, 2001. Propagules in dry wetland sediments as indicators of ecological health: effects of salinity. Regulated Rivers: Research and Management 17: 191–197.

StatSoft, 2001. STATISTICA System Reference. Statsoft, Tulsa, Oklahoma.

Stelfox, L., 2004. Assessment of potential contamination from the Moore River. Salinity and Land Use Impact Series Report SLUI33. Department of Environment, Perth.

Stone, C., T. Wardlaw, R. Floyd, A. Carnegie, R. Wylie & D. de Little, 2003. Harmonisation of methods for the assessment and reporting of forest health in Australia—a starting point. Australian Forestry 66: 233–245.

Teakle, L. J. H. & G. H. Burvill, 1938. The movement of soluble salts in soils under light rainfall conditions. Journal of Agriculture of Western Australia 15: 218–245.

Thomsen, J. B., 1999. Looking for hotspots. World Conservation 99: 6–7.

Walters, C., 1986. Adaptive Management of Renewable Resources. MacMillan, New York.

Webster, K. E., P. A. Soranno, S. B. Baines, T. K. Kratz, C. J. Bowser, P. J. Dillon, P. Campbell, E. J. Fee & R. E Hecky, 2000. Structuring features of lake districts: Landscape controls on lake chemical responses to drought. Freshwater Biology 43: 499–515.

Williams, W. D., 1987. Salinization of rivers and streams: An important environmental hazard. Ambio 16: 180–185.

Williams, W. D., 1999. Salinization: a major threat to water resources in the arid and semi-arid regions of the world. Lakes & Reservoirs: Research and Management 4: 85–91.

Wood, W. E., 1924. Increase of salt in soil and streams following the destruction of native vegetation. Journal of the Royal Society of Western Australia 10: 35–48.

Yesertener, C., D. P. Commander & P. D. Muirden, 2000. Groundwater and surface water outflow from the Yarra Yarra Lakes, Western Australia. In Hydro 2000, Interactive Hydrology, 3rd International Hydrology and Water Resources Symposium. Institute of Engineers, Australia, Perth, 323–328.