# State Salinity Strategy Wetland Biodiversity Monitoring Report:

# Lake Eganu 1998 to 2007

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*Our environment, our future*

# Table of Contents

**SUMMARY**

Lake Eganu is situated in the Pinjarrega Nature Reserve 220 km north of Perth. The wetland has suffered from increasing salinisation since the 1960s with up to 80% of invertebrate species and lake floor vegetation already lost and riparian vegetation reduced to a narrow band at the upper limit of filling.

Lake Eganu was included in the State Salinity Strategy Wetland Monitoring Program as an example of a secondarily salinised wetland with a recognised value for waterbirds and a long history of depth gauging and study. Data collection under the monitoring program commenced in 1998 and is on going. Data from 1998-2007 are presented for three components of biodiversity; vegetation, waterbirds and aquatic invertebrates.

Seventeen species of waterbirds and 38 invertebrate taxa were collected during monitoring. Eleven waterbird species recorded in the 1980s were not collected during monitoring while 3 saline specialists recorded during monitoring have not been reported previously. One waterbird species bred on the lake during monitoring; a significant reduction from 8 species recorded breeding in the 1980s. All waterbirds and invertebrates were of ubiquitous species with broad environmental tolerances.

Despite the continuing effects of salinisation, salinity is sufficiently reduced when the wetland fills for the water body to support significant waterbird abundance, with counts of 5000 birds during this study equally those recorded in the 1980s and 1990s. During these periods of lower salinity the wetland remains an important refuge for moulting Australian Shelduck. Similarly, invertebrate diversity falls within a range that indicates moderate condition during these periods of lake fill.

By contrast, lake filling and subsequent inundation of riparian vegetation and reduced depth to the water table exacerbate the declining health of vegetation of the riparian zone. This vegetation has continued to decline with no recruitment observed during the period of monitoring. There are likely to be follow-on effects if riparian vegetation declines further. Waterbird breeding success, in particular, is likely to be linked with an intact riparian zone.

## BACKGROUND

The Western Australian Salinity Action Plan (Anon 1996) was developed as a blue print for government action, in partnership with the community, to address the problems of landscape salinisation. The plan included strategies to manage the impact of salinity on natural (biological and physical) diversity and identified the need to monitor biodiversity in wetlands as a means of evaluating the achievement of the biodiversity conservation goals under taken as part of the plan. To this end the Department of Environment and Conservation (as its predecessor CALM) was charged with the responsibility to "…monitor a sample of wetlands, and their associated flora and fauna throughout the south-west to determine long-term trends in natural diversity and provide a sound basis for corrective action."

The Department of Environment and Conservation (and its predecessors) had been monitoring salinity and depth in over 100 wetlands since the 1970s (Lane and Munro 1983) and revitalised this program as the South West Wetland Monitoring Program (SWWMP - Lane *et al.* 2008) to partially meet the requirements of the Salinity Action Plan. As an extension of SWWMP the State Salinity Strategy Wetlands Biodiversity Monitoring program was commenced in 1997 to monitor waterbirds, invertebrates and flora in a sub-set of SWWMP wetlands. The Wheatbelt Wetlands Biodiversity Monitoring program commenced with a pilot study of 5 wetlands (Halse *et al.* 2002) and over the course of 1998 and 1999 a further twenty wetlands were added. These wetlands were selected according to a number of criteria (Table 1; see also Cale *et al.* 2004) enabling the relatively small sample of wetlands to be representative of the wide range of wetland types occurring in the region and to make best use of pre-existing knowledge.

The stated aims of the Wheatbelt Wetlands Biodiversity Monitoring program are encapsulated in the original action statement from the Salinity Action Plan, i.e.;

> *"To monitor a sample of wetlands, and their associated flora and fauna to determine long-term trends in natural diversity and provide a sound basis for corrective action."*

Each wetland in the program is sampled every second year to determine the composition of invertebrate and waterbird communities and every third year to assess the health and composition of vegetation communities. These data for biodiversity are comprehensive. Invertebrates from a very broad suite of taxa are identified to species level and complete counts of waterbirds are conducted three times in a monitoring year. Vegetation is monitored in set quadrats enabling the assessment of health in marked specimens of a wide range of species. To aid interpretation of biological data, data are also collected for wetland surface water chemistry, and from shallow monitoring bores. A detailed description of the monitoring protocol is given by Cale *et al.* (2004) and Froend *et al.* (1998) and an analysis of its efficacy was presented by Halse *et al.* (2000). The data collection regime is ongoing and the earliest sampled wetlands (i.e. those commenced in 1997) have now been sampled up to six times.

This report is one of a series of a series that will analyse and interpret the data collected for a single wetland within the Wheatbelt Wetlands Biodiversity Monitoring Program. These reports have been produced independently in the interest of decreasing the time taken for reporting.

*Table 1. Wheatbelt Wetlands Biodiversity Monitoring Program wetlands and selection criteria. 1 Monitoring design; wetlands from each of primary saline, secondarily saline, fresh, declining and improving, 2 Wetland listed in SAP (Anon 1996), 3 High conservation value, 4 Geographic representativeness, 5 Long record of data, 6 Management in catchment, 7 Size; very large wetlands were avoided except to meet other criteria.*

| **Wetland** | 1. Design | 2. SAP listing | 3. Conservation value | 4. Representative | 5. Data record | 6. Management | 7. Size |
|---|---|---|---|---|---|---|---|
| Altham | Primary saline | | √ | √ | √ | | √ |
| Ardath | Declining | | √ | √ | | | √ |
| Bennetts | Primary saline | | √ | √ | √ | | √ |
| Blue Gum | Fresh | | | √ | √ | | √ |
| Bryde | Improving | √ | √ | √ | √ | √ | √ |
| Campion | Primary saline | | √ | √ | √ | | |
| Coomalbidgup | Declining | | √ | √ | √ | | √ |
| Coomelberrup | Declining | | | √ | √ | √ | √ |
| Coyrecup | Secondary saline | | √ | √ | √ | √ | |
| Dumbleyung | Secondary saline | | √ | √ | √ | | |
| Eganu | Secondary saline | | √ | √ | √ | | √ |
| Fraser | Fresh | | √ | | | | √ |
| Goonaping | Fresh | | √ | | √ | | √ |
| Kulicup | Fresh | | √ | √ | √ | | √ |
| Logue | Fresh | | √ | √ | √ | √ | √ |
| Noobijup | Improving | √ | √ | √ | √ | √ | √ |
| Paperbark | Fresh | | √ | √ | √ | | √ |
| Parkeyerring | Secondary saline | | √ | √ | √ | | |
| Pleasant View | Fresh | | √ | √ | √ | | √ |
| Ronnerup | Primary saline | | √ | √ | | | √ |
| Toolibin | Improving | √ | √ | | √ | √ | √ |
| Towerrining | Improving | | √ | | √ | √ | |
| Walyormouring | Secondary saline | | √ | √ | √ | | |
| Wheatfield | Primary saline | √ | √ | √ | | √ | |
| Yaalup | Declining | | √ | √ | √ | | √ |

## INTRODUCTION

Lake Eganu is a secondarily salinised wetland located 220 km north of Perth (30°0'7''S, 115°52'26'') within the Pinjarrega Nature Reserve (4,686ha) which adjoins the Capamauro Nature Reserve (4,710ha) to the north (Fig. 1). Both reserves are uncleared but have not protected Lake Eganu from salinisation because most of the lakes inflows come from cleared parts of the catchment outside these reserves (Lyons *et al.* 2007). The lake has a well defined basin with an area of 110 ha (Lyons *et al.* 2007) and is part of a suite of connected wetlands forming the Coonderoo River between Yarra Yarra Lake upstream and Lake Pinjarrega downstream (Fig 1). Little is known about the history of Yarra Yarra Lake, although it possibly overflowed in 1917, and perhaps 1918, into the Coonderoo River and subsequently Lake Eganu, (Yesertener *et al.* 2000). The bulk (70%) of surface water inflow to Lake Eganu comes from the Marchagee Tributary which drains a 430 000 ha catchment to the east and south east and enters the Coonderoo River immediately north of Lake Eganu. Eighty seven percent of this catchment is cleared with most clearing having been completed by the early 1960s, with valley floor clearing likely to have been very extensive by as early as the late 1920s. Base flows in the Marchagee Tributary have a salinity of 35-50 g/L (Lyons *et al.* 2007).

Upstream of Lake Eganu, groundwater from Yarra Yarra Lake discharges into the Coonderoo River with an approximate annual discharge of 0.3 M $m^3$/yr. The point at which the Coonderoo switches from a gaining river to a losing river with respect to groundwater occurs upstream, in the vicinity of Lake Eganu. Lake Eganu has probably only recently become seasonally connected to the regional watertable (Stelfox 2004).

Lake Eganu has been gauged and water levels and salinity monitored since 1979 (Lane and Munro 1983). These data indicate a significant trend of increasing depth, but no significant trend of increasing salinity (Lane *et al.* 2004). Cale *et al.* (2004) however suggest that minimum salinities occurring under full or near full conditions (where concentration effects can be eliminated) in 1998 are greater than reported in the 1980s and this is confirmed by analyses of Lyons *et al.* (2007) that indicate for depths above 2m there is a significant trend of increasing salinity.

Vegetation of the wetland has been described (Halse *et al.* 1993, Gurner *et al.*1999) and an analysis of vegetation change based on aerial photography (Lyons *et al.* 2007) indicated that prior to salinisation the wetland was covered by an open-woodland dominated by *Casuarina obesa* and *Melaleuca strobophylla*. The health and distribution of trees has progressively declined so that a band <50 m wide remains at the upper limit of filling. A broader band (<200 m) of dead standing timber still covers much of the margin of the lake basin leaving a central open area (Fig. 2).

Waterbirds, aquatic invertebrates and fish have previously been collected from Lake Eganu. Waterbirds were intensively surveyed between 1981 and 1985 (Jaensch *et al.* 1988) with a total of 24 species, including 8 breeding species, recorded from 19 surveys. Using waterbird surveys from 1999-2004, Lyons *et al.* (2007) reported a decline in waterbird species richness accompanied by the loss of 9 fresh to hyposaline species and the appearance of 3 saline species. Aquatic invertebrates have been documented (Halse 1981, Cale *et al.* 2004, Lyons *et al.* 2007) and indicate a fauna typical of salinised wetlands. Lyons *et al.* (2007) suggest that increasing salinity has reduced invertebrate species richness by 80% since the 1960s. The fish *Pseudogobius olorum* (Swan River goby) was collected by Halse (1981), but does not appear to have been collected since.

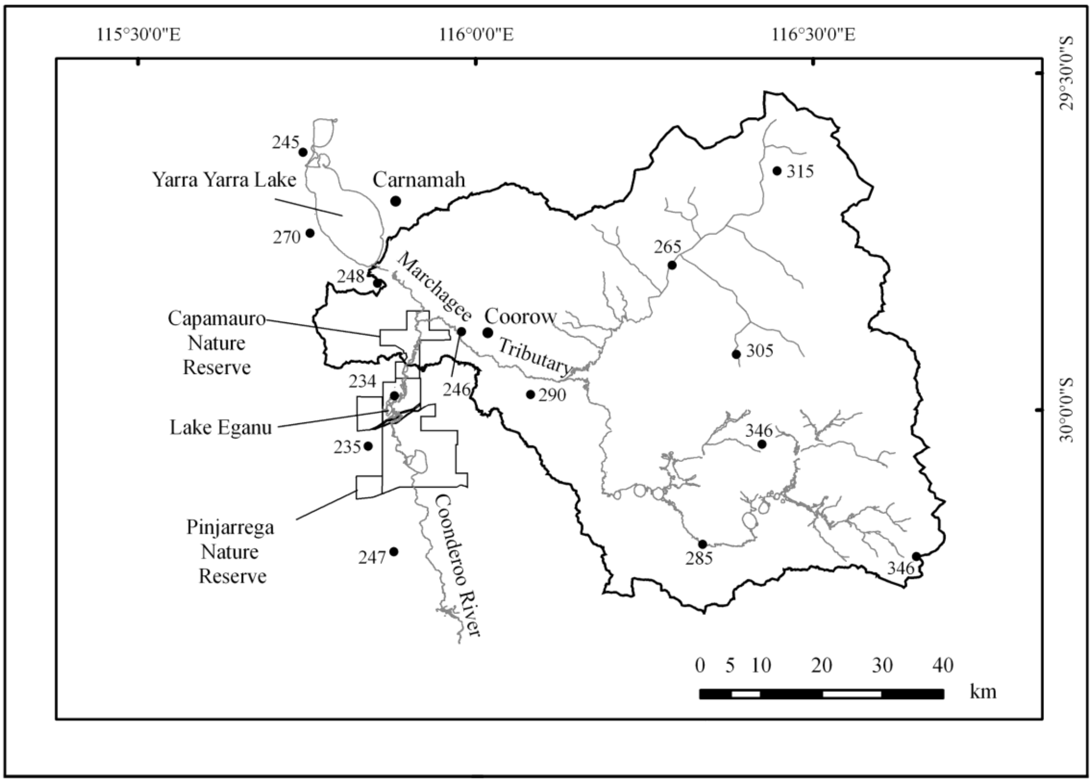


*Figure 1. Catchment setting of Lake Eganu. Numbers indicate spot heights in metres (adapted from Lyons et al. 2007).*

## METHODS

The data reported here was collected as part of the Wheatbelt Wetland Biodiversity Monitoring Program. The protocol for this program is described briefly below and in more detail by Cale *et al.* (2004). Vegetation transect design is described in detail by Gurner *et al.* (1999).

### Study Sites

*Invertebrates and waterbirds*

Invertebrate sampling was conducted at two sub-sites within the wetland with the intention of increasing the total number of species collected (Fig. 2). Sub-site A is in the vicinity of the depth gauge on the lake's eastern shore, while site B, selected to sample a different quadrant of the wetland, lies in an embayment on the opposite shore. The actual location of sampling sites varies from year to year according to water levels. Sub-sites have no relevance to waterbird data which were collected over the whole wetland.

*Vegetation*

Two transects were positioned on the eastern side of the lake to sample representative stands of trees and shrubs dominated by *Casuarina obesa*. These transects sampled the upper riparian zone and were orientated at right angles to the prevailing slope and correspondingly to the wetland edge. Both transects were dominated by a combination of bare saline clays variably overlain by low sandy dunes closer to the lake edge. Transect 3 occurred at a subsidiary wetland and sampled a mixed stand of *C. obesa* and *M. strobophylla*. (Fig. 2). This wetland has not experienced significant vegetation decline and was sampled to provide a comparison to the main lake. Transect horizontal and vertical locations were fixed using RTK

GPS relative to the depth gauge benchmark (BM) (Table 2, Fig. 2). Since no state survey mark (SSM), with known MGA 94 values was readily available, values in Table 2 are approximate MGA 94 (+/- 10m). In the future a block shift of horizontal values could be performed by connecting the BM to the nearest available SSM. Vertical values could similarly be converted to the Australian Height Datum. The adjusted mCALM values (Table 2) however are *true* relative to the lowest point of the lake (i.e. equivalent to water depth measurements).

*Table 2. Location of vegetation transects and groundwater monitoring bores. Transect fixes are for the entire transect corner pegs. mCALM values are the vertical height above the lowest point of the lake. Horizontal values (eastings and northings) are approximate MGA 94 (+/- 10m). Adjusted vertical mCALM values are true heights above the lowest point of the lake.*

| **Feature** | **Northing (MGA94) Zone 50** | **Easting (MGA94) Zone 50** | **mCALM** |
|---|---|---|---|
| **Transect 1** | | | |
| | 6680894.429 | 391559.329 | 2.94 |
| | 6680908.507 | 391617.278 | 3.79 |
| | 6680927.746 | 391612.919 | 3.66 |
| | 6680913.818 | 391554.597 | 2.89 |
| BORE 1 | 6680888.679 | 391580.568 | 3.04 |
| BORE 2 | 6680904.907 | 391618.132 | 3.81 |
| | | | |
| **Transect 2** | | | |
| | 6680416.766 | 391428.815 | 2.41 |
| | 6680406.002 | 391487.89 | 2.52 |
| | 6680386.516 | 391484.231 | 2.71 |
| | 6680397.141 | 391425.374 | 2.31 |
| BORE 1 | 6680409.256 | 391490.038 | 2.53 |
| BORE 2 | 6680417.161 | 391459.85 | 2.43 |
| | | | |
| **Transect 3** | | | |
| | 6683522.294 | 393599.602 | 3.12 |
| | 6683532.402 | 393560.808 | 3.14 |
| | 6683513.172 | 393555.631 | 3.3 |
| | 6683503.09 | 393594.329 | 3.35 |
| BORE 1 | 6683530.672 | 393601.299 | 3.13 |
| BORE 2 | 6683533.673 | 393583.859 | 3.12 |

**Timing of sampling**

*Invertebrates and waterbirds*

Lake Eganu has been sampled every second year from 1998 to the present. Three successive sampling events occur each sampling year. Notionally, these three events are "Late-winter" (Aug-Sep), when inflows are occurring or reaching their annual conclusion; "Spring" (Oct-Nov) timed to coincide with substantial development of invertebrate communities and "Autumn" (Mar-Apr) when wetlands are at their lowest water levels and salinised wetlands are likely to be stressed. These sampling periods span two calendar years. For clarity of discussion a monitoring year has been defined to span from Late-winter to autumn and is labelled by the year of the Late-winter and spring samples. Frequently, some wetlands will fail to fit neatly with the conceptual sampling regime in some years because either Summer rainfall results in peak wetland volumes after the Spring sampling or winter inflows fail to occur. Consequently, samples from any sampling year represent a snap shot of conditions at some unknown point along the trajectory of a hydrological cycle. However, this was not the case at Lake Eganu which showed a consistent seasonal hydrocycle during the study period.

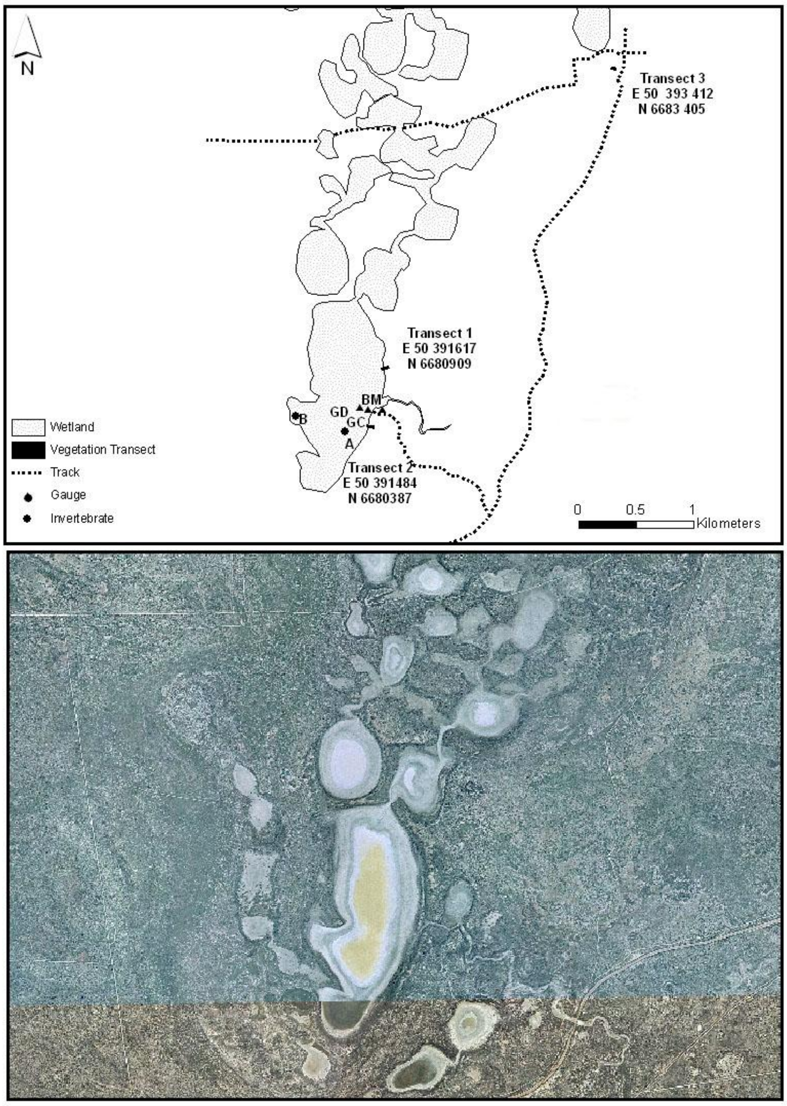


*Figure 2. Location of sampling sites for Lake Eganu. (GC - depth gauge C, GD –depth gauge D, A – invertebrate sub-site A, B – invertebrate sub-site B, BM – survey bench mark). Vegetation transect coordinates are for the upper left corner peg (as seen when facing the wetland). Recent aerial photography shown in the lower panel highlights the absence of vegetation across the lake basin.*

*Vegetation, shallow groundwater and soil conductivity.*

Sampling of the vegetation commenced in 1998, with sampling repeated in 2001, 2004 and 2007. Field measurements occurred in November or December in each of the four sampling years, with soil electrical conductivity measurements made at the same time as the vegetation sampling. Late spring – early summer measurements were considered valid since soil moisture was still likely to be sufficient (>20%) below the surface to provide meaningful conductivity measurements. Shallow groundwater monitoring at each transect commenced in December 1999 with two measurements per year. Depth to groundwater measurements were timed to maximize the probability of capturing seasonal maxima and minima (typically March and September).

**Sampling methods**

*Water Chemistry*

A range of water chemistry parameters were measured in order to assist the interpretation of waterbird and invertebrate data (Table 3). Several parameters were measured at sub-site A on all sampling occasions: 1) Depth was measured at permanent gauges installed as part of the South-West Wetland Monitoring Program (Lane *et al.* 2004), 2) electrical conductivity (EC) and pH were measured using a WTW 340i hand-held meter, 3) chlorophyll (sum of chlorophyll a, b and c) and phaeophytin concentration was determined in the laboratory (WA Chemistry Centre method iCHLA1WAC) from a known water volume (usually 1L) filtered through a GMC glass fibre filter paper, 4) a water sample filtered through 0.45 $\mu$m acetate syringe filter was collected for determination of total filtered nitrogen (TFN) and total filtered phosphorus (TFP). During Spring sampling these data are also collected at sub-site B.

During Spring sampling, un-filtered water samples from sub-site A were used for the laboratory determination of ionic composition, total dissolved solids (TDS), turbidity, colour, alkalinity and hardness (WA Chemistry Centre methods iMET1WCICP, iSOL1WDGR, iTURB1WCZ, iCOL1WACO, iALK1WATI, iHTOT2WAC respectively).

*Table 3. Frequency and location of water chemistry measurements. See text for description of parameters.*

| Parameter | Late-winter Site A | Spring Site A | Spring Site B | Autumn Site A |
|---|---|---|---|---|
| depth | √ | √ | | √ |
| temp | √ | √ | √ | √ |
| EC | √ | √ | √ | √ |
| pH | √ | √ | √ | √ |
| TFN/TFP | √ | √ | √ | √ |
| chlorophyll | √ | √ | √ | √ |
| ionic composition | | √ | | |
| TDS, turbidity, colour hardness alkalinity | | √ | | |

*Waterbird Census*

Waterbirds were surveyed on all sampling occasions within a monitoring year. All birds of all obligate wetland species (includes species such as Swamp Harrier) are identified and counted using binoculars or a spotting scope. As much of the wetland as was practical is traversed; by boat when water levels exceeded 60cm or on foot at lower depths. When water levels were high, channel areas at the north end of the wetland were flooded and consequently included in the survey.

*Invertebrate sampling*

The invertebrate sampling protocol focuses on maximising species richness. This protocol has been documented extensively and used in a range of biodiversity survey and monitoring studies within the wheatbelt region and elsewhere in Western Australia (Halse *et al.* 2002, Cale *et al.* 2004, Pinder *et al.* 2004, Cale 2007, Lyons *et al.* 2007).

An invertebrate sample comprises two sub-samples collected using D-framed pond nets with intermediate (250 μm) and fine (50 μm) mesh sizes. The two sub-samples are collected in order to increase the efficiency with which both macro- and microinvertebrates are collected. While more difficult to identify, microinvertebrates have been shown to comprise an average of 45% of collected species in wheatbelt wetlands (Halse *et al.* 2002, Cale 2007). Each sub-sample was collected over a large area including all identifiable habitats (50 m sample path within a 200m radius boundary). The fine mesh sub-sample was collected first to reduce disturbance and did not sample benthic sediments to reduce fouling of the mesh. The 250 μm mesh net is used to sample all habitats, including stirred up sediment, organic litter and macrophytes. During laboratory sorting specimens from these two sub-samples were combined to produce a single sample for identifications.

An invertebrate sample (comprising the two sub-samples) was collected for each sub-site to yield a pair of samples for the wetland. Data from these samples were recorded separately although analyses reported here utilise combined data. Halse *et al.* (2002) concluded that about 75% of invertebrate taxa present at a wetland were collected by paired samples. Studies in the Pilbara (Pinder unpublished data) suggest that a single sample collects about 70 to 80% of the species present at one sub-site.

Invertebrate samples were processed to retrieve as many species as possible and specimens were identified to the lowest taxon possible. Currently, several Dipteran families (Dolichopodae, Tabanidae, Tipulidae and Muscidae) are identified to family level only and Turbellaria, Nematoda, Mestostigmata and Oribatida are not determined beyond these nominal taxa. A voucher collection using National Register of Taxa Codes (Environmental Monitoring Unit, Environmental Protection Authority Victoria) is maintained at the Wildlife Research Centre, Woodvale and used to maintain consistency of identifications. A relational database with a master taxonomic list enables sample invertebrate data to be updated following taxonomic revisions.

While all specimens were identified to the lowest level possible the maturity or gender of specimens sometimes prevented identification of some taxa to species level. Within a sample these taxa do not impair the calculation of species richness for comparison between samples. However, when multiple samples (dates) were to be compared, e.g. during multivariate analyses, it was necessary to adjust species lists so that identifications at different taxonomic levels did not add spurious taxa to the analysis. This was achieved by deleting or combining taxa so as to lose as little information from the dataset as possible. For example the presence of *Berosus* sp. and *Berosus munitipennis* at different wetlands/dates within a dataset would be resolved to *Berosus* sp. across the dataset by combining the two taxa.

Species and family richness, and guild composition were calculated using species lists corrected for both sub-sites on a sampling date within the wetland. Richness values are plotted against the reference ranges (Table 4) used by Sim *et al.* (2008) as part of a trial of national indicators of wetland condition and originally developed by Jones *et al.* (2008).

Invertebrate assemblages, or groups of co-occurring taxa, were developed by Pinder *et al.* (2004) from their observed patterns of association in wheatbelt wetlands. Some of these assemblages were associated with particular wetland types while others were widespread, their richness within a wetland being best predicted by salinity and climate variables (Pinder *et al*. 2004). These assemblages were used in this study as a tool for describing invertebrate community composition. Where species were collected that did not appear in the study of Pinder *et al.* (2004) they were assigned to an undefined assemblage labelled U. The undefined assemblage does not imply any association between members, but rather a lack of information about their associations.

*Table 4. Taxonomic richness reference ranges (after Sim et al. 2008)*

| | | High condition | Medium condition | Low condition |
|---|---|---|---|---|
| Species | | >48 | 25-48 | 24 |
| Family | | >33 | 19-33 | 18 |

*Vegetation condition and population structure*

The detailed vegetation methods outlined in Lyons *et al*. (2007) are largely restated here. Transects 1 and 2 consisted of three contiguous 20 x 20 m plots, with transect 3 at the subsidiary wetland made up of two plots for a total length of 40 metres. Each 20 x 20 m was subdivided into five 4 x 20 m subplots orientated parallel with the wetland edge. 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 plants basal area was calculated as the sum of the five cross sectional areas. Diameter at breast height (DBH) was not measured for stems with DBH < 0.02 m. The number of seedlings/saplings (largest stem <0.02 m DBH) of overstorey taxa were counted within each subplot.

For each plant, an assessment of crown condition was made using a visual scoring system based on the original scheme of Grimes (1978) and adopting the modifications of Stone *et al.* (2003). In the current study three components (crown density, dead branches and epicormic growth) were scored and summed to give an aggregate condition score. Both crown density and dead branches were scored on a five point scale with values of 9, 7, 5, 3, and 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.

At each transect changes in basal area of the dominant overstorey taxa, *Casuarina obesa* and *Melaleuca strobophylla* were calculated for the period 1998 to 2007 at the subplot scale (4 x 20 m) for plants >20mm DBH. Size class distributions were plotted for each species at each transect for the four sample years and included seedlings measuring less than 20mm DBH. Plots were examined for changes in population structure and recruitment although size classes could not be directly equated to age structure.

Elevation profiles were determined every 4 m along both sides and the centre of transects using a dumpy level and staff. For transects 1 and 2 the data was transformed to height above the lowest point of the wetland. These data enable the examination of the inundation history of transects during wetland fill events by linking them to wetland depth gauge data (data incorporated from Lyons *et al.* 2007). The elevation of transect 3 is arbitrary and cannot be directly related to Lake Eganu water depth.

*Vegetation transects -shallow groundwater monitoring*

Shallow groundwater monitoring bores were established adjacent to vegetation transects. Two bores were installed beside each transect approximately 10-15m towards the transect centre from the upslope and downslope ends. Observation bores were constructed of 45 mm diameter PVC pipe with the lower section slotted and the bore hole sealed with blue metal and bentonite clay. Bore depths ranged from 2.4 m to 3.8 m. Depth to groundwater below local ground level was measured using an electrical conductivity dipper tape and salinity (as electrical conductivity, mS $m^{-1}$) and pH were measured using a TPS-W81 meter.

*Vegetation transects - soil electrical conductivity*

Measurements of soil apparent electrical conductivity ($EC_a$ mS/$m^{-1}$) were made at each vegetation sampling event using a handheld EM38 (Geonics Limited) with data collected in the horizontal (HD) and vertical dipole (VD) orientations. Penetration in these orientations is approximately 0.75 m and 1.5 m (80% response) respectively. Measurements were conducted on a fixed grid of points, every 4m along the length of the sides and centre line of transects and repeated each sampling year. The primary focus of EM38 measurements was to detect major shifts in soil conductivity both spatially and over time within transects. Care was taken to avoid interference from the metal pickets marking quadrats by relocating the instrument approximately one metre inside transects at peg locations.

Given the saline soils around the lake, $EC_a$ data were examined without calibration. This was deemed appropriate given that for such soils the conductivity is dominated by the salt concentration rather than soil texture or cation exchange capacity.

**Invertebrate and waterbird data analysis**

*Multivariate ordinations*

All ordinations were performed using the Semi-Strong Hybrid (SSH) Multi Dimensional Scaling (MDS) algorithm provided in the PATN package (PATN version 3.12 Belbin and Collins 2008). The Bray-Curtis measure of association was used for presence/absence data and the Gower metric for abundance data. Ordination axes were not rotated. Minimum Spanning Tree networks were applied to ordination plots (using the MST function in the PATN package) to facilitate interpretation of three dimensional plots. MST is the shortest network that connects all sample points with only a single pathway between any two sample points. Flexible UPGMA classification (PATN) was used to determine group membership of samples after ordination with the number of groups derived equal to the square root of the number of samples in the ordination.

Annual waterbird community structure (i.e. species found at sometime during the monitoring year) was combined with data from historical surveys and data from marker wetlands for analysis. Historical data for Lake Eganu was collected between 1980 and 1985 (Jaensch *et al.* 1988). Only data from years with 3 surveys from comparable seasons were included. The five marker wetlands and the rationale for their inclusion in the analysis are described in Cale *et al.* (2004). Each marker wetland is considered typical of a wetland type. Lake Toolibin (data collected in 1983 by Jaensch *et al.* 1988) is an example of high quality waterbird habitat in an open wooded fresh to brackish inland wetland. Lake Pleasant View (data from 1998; Cale *et al.* 2004) is typical of reed swamps, with both low diversity and abundance of species associated with freshwaters. Lake Pinjarrega (data collected in 1983 by Jaensch *et al.* 1988) has a diverse waterbird fauna and is an example of a deeper brackish wetland with a recent history of secondary salinisation. Lake Goorly (data from a single survey in October 1999, DEC unpublished data) had a fauna typical of saline playas. Lake Altham (data from 1998; Cale *et al.* 2004) is typical of the species poor community present at hypersaline wetlands.

*PCC analyses of environmental parameters*
Principal Component Correlation (PCC) analyses were conducted using the PATN software package (Belbin and Collins 2008). For waterbirds a PCC analysis was conducted using only water chemistry parameters from site A that were collected on all sampling dates (Table 3) and an SSH ordination of the individual waterbird survey lists (i.e. Late-winter, Spring, Autumn). Ordinations were conducted on both presence/absence and abundance data with and without masking of singleton species (i.e. those collected only once). Abundance data without masking was more informative and is presented here.

For invertebrates a PCC analysis was conducted using Late-winter and Spring values of parameters collected at site A including those only collected in Spring (Table 3) and an SSH ordination of the annual species presence/absence.

All concentrations of anions and cations were re-calculated as milli-equivalents per litre. Normality of parameters was tested using the Shapiro-Wilkes (W) test and $\alpha = 0.05$ and data were transformed as required to approximate normality. Transformed data were plotted and parameters that did not possess at least three value levels were excluded from analysis to avoid two point correlations.

Following PCC analysis “significant vectors” were determined using the MCAO (Monte-Carlo Attributes of Ordination) module of PATN. Vectors with a correlation greater than all but 5% of randomised permutations were considered significant.

**Vegetation data analysis**

Three indices of vegetation condition were examined to assess the “health” of the riparian plant communities of the lake. 1) Changes in stand basal area over time. 2) Changes in canopy health rating over time. 3) Examination of the size class distribution (as a surrogate for age structure) of stands and an assessment of recruitment.

Percentage changes in basal area from 1998 to 2007 were calculated for each dominant overstorey species at transects at the sub-plot scale, and plotted against position along transects and in relation to the transect elevation profiles. The decline or accumulation of basal area (decline – due to death, accumulation due to growth and/or recruitment) was also examined by linear regression against apparent soil conductivity.

Stand canopy condition was examined over time by comparison of aggregate canopy condition scores at each sample time. Two approaches to the data were taken. The mean condition of trees *extant* at all sample time were plotted over time (i.e. are surviving trees showing significant differences in condition between years), and secondly the condition of the *stand* over time was examined by assigning trees that died during the study a condition score of zero and retaining them in the analysis. Freidman non-parametric ANOVA (Statsoft, 2001) was applied to repeated measures data for both compilations for *C. obesa* and *M. strobophylla* at the transect scale. Wilcoxon matched pairs test Statsoft, 2001) for dependent samples was also performed, comparing 1998 to 2007 data.

## RESULTS

### Water chemistry

Depth data suggest that the lake fills to at least 2 m, at the depth gauge, in most years; generally as a result of winter rainfall (Fig. 3a). After filling water levels recede through groundwater recharge or evaporation resulting in either the lake drying or nearly drying as in 2000 and 2004 (Fig. 3b) or retaining moderate depth until refilling the following winter as in 1998. The lake dried at some time during the 2000, 2002 and 2006 monitoring years (Fig. 3b). The highest recorded depth was 2.87 m in September 1999 (Lane *et al.* 2004) when the region was extensively flooded. At this time the lake is known to have overflowed with extensive flooding recorded at the Greenhead Road immediately south of the wetland.

A discrepancy in depths recorded at the gauge was first observed in 2000, with lake depth actually 0cm at 60cm on the gauge. It is likely that the formation of a thick salt crust as the lake evaporates accounts for at least 20cm of this discrepancy, while the remaining discrepancy is the result of the gauge being surveyed to the deepest point in the lake which occupies a very small area of the lake bed. When the lake was dry depth was recorded as 0 m but for shallow depths, 1 - 25 cm, depth was recorded from the gauge. Because lake depth was not verified at greater depths and the extent of error is unknown the depth at the gauge is reported here to enable comparison with already published datasets (e.g. Lane and Munro 1983 and Lane *et al.* 2004).

Across all samples ($n = 11$ between Aug 1998- Oct 2006) EC and depth at gauge were negatively correlated ($r = -0.98$, $p<0.05$, see also Fig 4). Similarly, total filtered nitrogen, chlorophyll and temperature were negatively related to depth at gauge ($r<-0.58$, $p<0.05$ $n = 11$) and positively correlated with EC ($r>0.57$, $p<0.05$, $n = 11$). Depth at gauge and pH were positively correlated ($r = 0.82$, $p<0.05$) reflecting an equally negative correlation of pH with EC i.e. as the lake becomes more saline it also becomes more acidic.

Total nitrogen concentration tended to increase throughout each year as the wetland dried. Average late winter concentration was 3.5 mg/L ($n = 12$, SD = 2.99) while average Spring concentration was 6.9 mg/L ($n = 12$, SD = 7.60). The lake was often dry for autumn sampling but average concentration from available data is 7.3 mg/L ($n = 4$, SD = 4.22). A minimum concentration of 0.34 mg/L was recorded where depth at gauge was in excess of 2m while the highest concentration of 22 mg/L was recorded when the lake was reduced to a depth of 9cm (i.e. 0.79 m at the gauge). Total soluble phosphorus was at low concentration (<10 μg/L) on most occasions with higher values (max = 330 μg/L) only recorded at very low depths (<0.79 m at the gauge).

Salinity (electrical conductivity) ranged from 25.1 mS/cm at 2.44 m depth at gauge (1998) to the highest recorded salinity of 236.0 mS/cm at a depth at gauge of 0.7 m (2002). At shallow depths the remaining water becomes super saturated with salts and the benthos is covered in salt crystals. While sodium was always the dominant cation, in dry years there was a tendency for ionic composition to change such that magnesium increases in proportion while calcium, which is never in high proportion, decreases.

Lake water was generally clear with turbidity ranging from 0.2-25 NTU, but greatly increased turbidity was observed in Nov 1998 (990 NTU). High water clarity allows dense stands of submerged macrophytes to dominate the benthos except at high salinities. The presence of macrophytes in Nov 1998 suggests the observed high turbidity was a short term event, possibly in response to wind induced wave action. No submerged macrophytes were observed at salinities above 128.6 mS/cm.

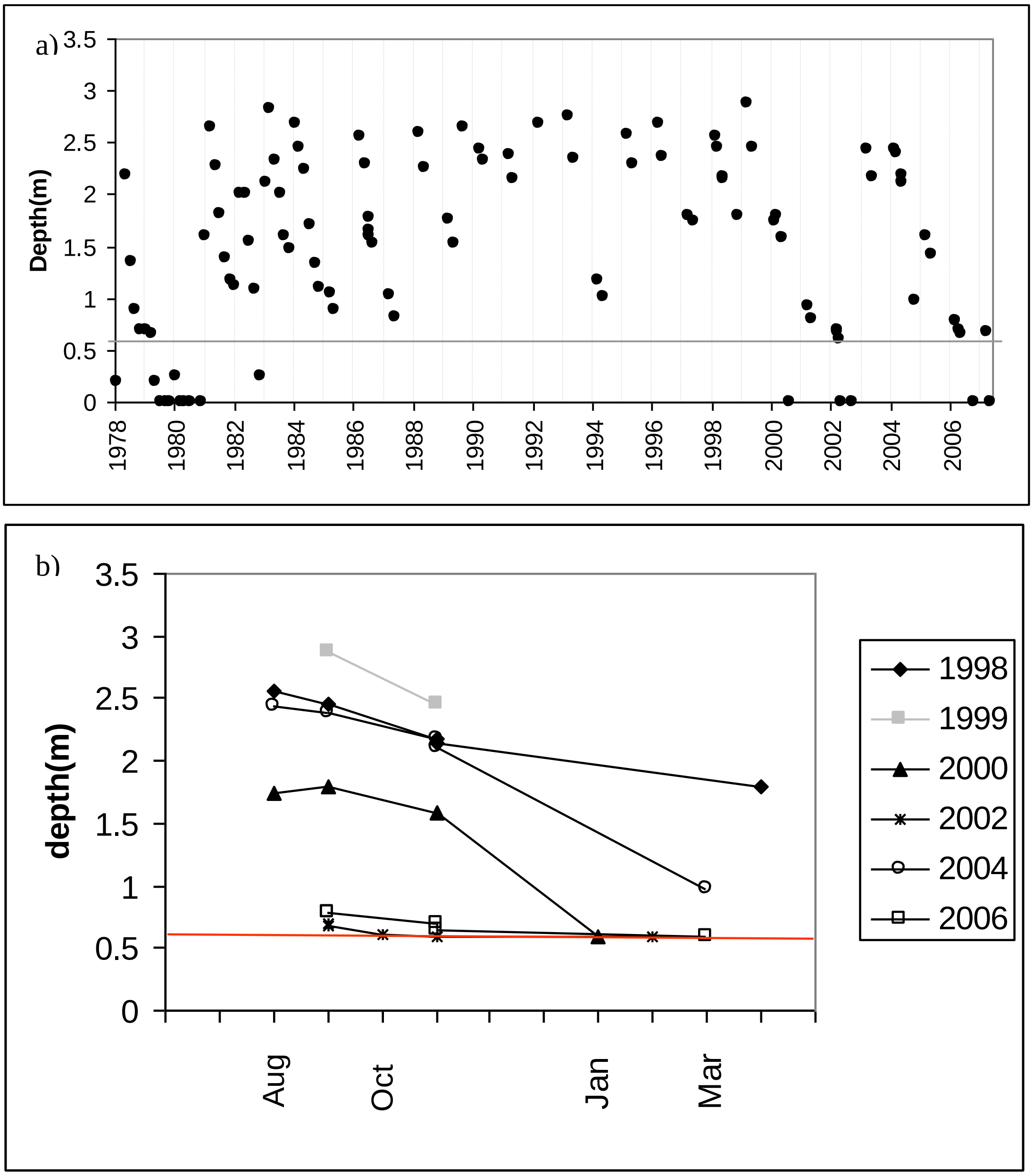


*Figure 3. Depth at gauge in Lake Eganu. Note that the portion of the wetland that is sampled in this study is dry at 0.6m on the gauge (horizontal line in figures). a) Monitoring program data plus long-term data provided by Jim Lane; Dept Env. & Cons., (see also Lane et al. 2004). Annual tick marks (vertical lines) are at June of each year .b) Depth at gauge for monitoring years showing pattern of filling and drying. Depth in 1999 represents a rare flooding event.*

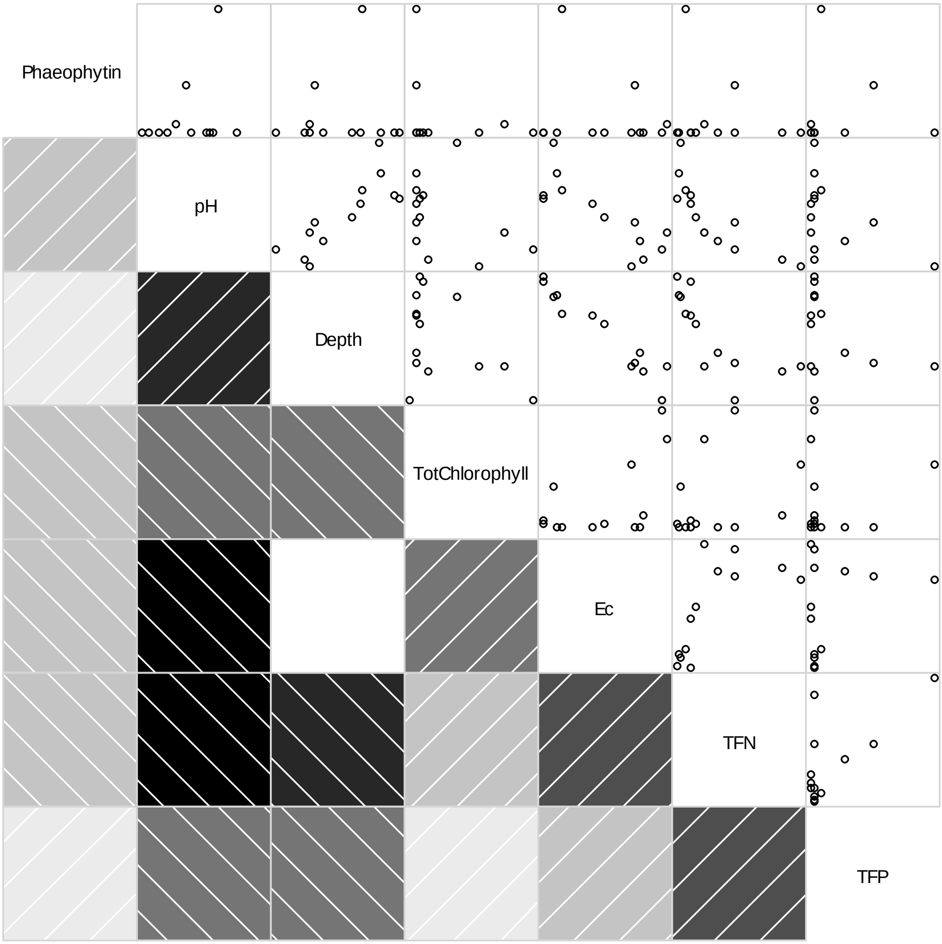


*Figure 4. Correlation matrix for seven water chemistry parameters (diagonal) collected on all biological sampling dates. Darkest colours (lower panel) represent largest r values, and data values are plotted in upper panel. Significant correlations are discussed in the text.*

## Waterbirds

*Description of waterbird assemblage*

The waterbird fauna at Lake Eganu was dominated by "swimming" species on most occasions, reflecting its value as a loafing and feeding wetland whenever sufficient water is present. Richness was 10 species or fewer (Fig. 5) on all occasions (mean = 4.5 for surveys where water or birds were present). Breeding has only been recorded on one occasion when broods of Grey Teal were observed. A total of 17 species have been recorded during the monitoring period (Table 5) and 8 of these are amongst the ten most commonly recorded species found by Jaensch *et al*. (1988) during extensive surveys of south-west Western Australia between 1981-1985. Using the categories of Jaensch *et al*. (1988), Lake Eganu has been of *low* importance to waterbirds, in terms of number of species and number of breeding species, and of *low* to *moderate* importance in terms of abundance in most years. In Nov 1998 abundances in excess of 5,000 birds indicated the lake was of *high* importance to waterbirds in that year.

*How waterbird assemblage changes over time*

From individual surveys, waterbird assemblages remained consistent in composition when lake depth was greater than 1m (Group 1 samples of Fig. 6.). When the lake was shallow, species composition was erratic (Group 2 samples of Fig. 6) with a general absence of several species of duck (e.g. Australasian Shoveler, Pacific Black Duck, Musk Duck and Hardhead), and the opportunistic use of the wetland by a number of wader and shore species, each sighted on only single occasions (see also Fig.7). The near dry conditions of Autumn 2000 (Group 3 sample of Fig.6) supported a single species only.

Annual waterbird assemblages were similar in 1998 and 2004 (Fig.8) when water levels were in excess of 2.4 m and represented a substantial portion of the lakes capacity. Species richness was highest in these years and guild structure was similar (Fig. 7). These sampling years are most similar to historical waterbird surveys (Jaensch *et al.* 1988), with the 1998 sample falling within the envelope of historical data. The 1998 and 2004 bird faunas are most similar to the marker assemblage of Lake Pinjarrega, although they are still less similar to this marker assemblage than they are to each other. Lake Pinjarrega was selected as a marker wetland representing a recent history of salinisation and displaying a diverse waterbird fauna thought typical of brackish deep water lakes. Lake Pinjarrega is geographically close to Lake Eganu and has a similar habitat structure and salinity, although it is much larger.

In 2002, when the lake was almost dry, only a single Banded Stilt was encountered. Consequently, the assemblage for 2002 does not match that of any other occasion and points to the low quality of the wetland for waterbirds when only a few centimetres deep and hypersaline. At moderate water levels (1.6-1.8 m) as in 2000 the waterbird assemblage had a larger proportion of wading and shore species (Fig. 7) and the richness of species of duck was lower.

Eleven species recorded by Jaensch *et al.* (1988) i.e., Freckled duck, Blue-billed duck, White-necked Heron, Australian Wood Duck, Little Pied Cormorant, Red-kneed Dotterel, Yellow Spoonbill, Black-winged Stilt, Whiskered Tern, Great Cormorant and Swamp Harrier have not been recorded during the current monitoring program. Conversely, three saline lake species recorded by us, Banded Stilt, Silver Gull and Red-capped Dotterel, had not been recorded prior to monitoring. The decline in species richness since the 1980's is strongly correlated with increasing salinity (Lyons *et al.* 2007).

*Factors associated with a change in waterbird assemblage*

There was a significant positive correlation between waterbird species richness and depth at gauge ($r = 0.92$, $df = 11$, $p<0.001$ and see Fig. 5). While depth at gauge was negatively correlated with a range of parameters (most notably EC) it is a useful indicator of environmental factors structuring waterbird communities at Lake Eganu. The low species richness observed during 2002 was a direct response to the near dry conditions at the lake. The suite of co-correlated parameters EC, pH, depth and Total Filtered Nitrogen (TFN) were the only PCC vectors significantly correlated with waterbird abundance and composition (Fig. 6). Group 1 samples of this ordination were collected at greater wetland depth and included more species and individuals than surveys at lower depths. It is not possible, with the data available, to separate the effects of individual parameters on waterbird community structure.

The number of species reliant on shore habitats (i.e. large and small wader and shore-feeder guilds) were recorded too infrequently to accurately model their relationship with lake depth; however, both richness and abundance appeared to be constrained by high depth at one extreme and high salinity at the other.

Grey Teal were one of the most frequently encountered species and generally comprised a large proportion of total waterbird abundance (Table 5). There was a significant correlation between the abundance of this species and depth at gauge (regression $r^2 = 0.40$, $F = 8.14$, df =

13, p<0.05) with no Grey Teal present below depths of 1.58m at the gauge (note however that there are no data for depths of 1.0 – 1.5 m). A similar dependence on lake depth can be inferred for the Australian Shelduck which was periodically in high abundance at depths greater than 1.58m. Australian Shelduck were most commonly encountered during Spring surveys when many individuals were observed to be moulting and are likely to be dependent on higher water levels to afford protection during this flightless period. Shelduck were not present in Spring surveys when lake depth at gauge was less than 1.58 m.

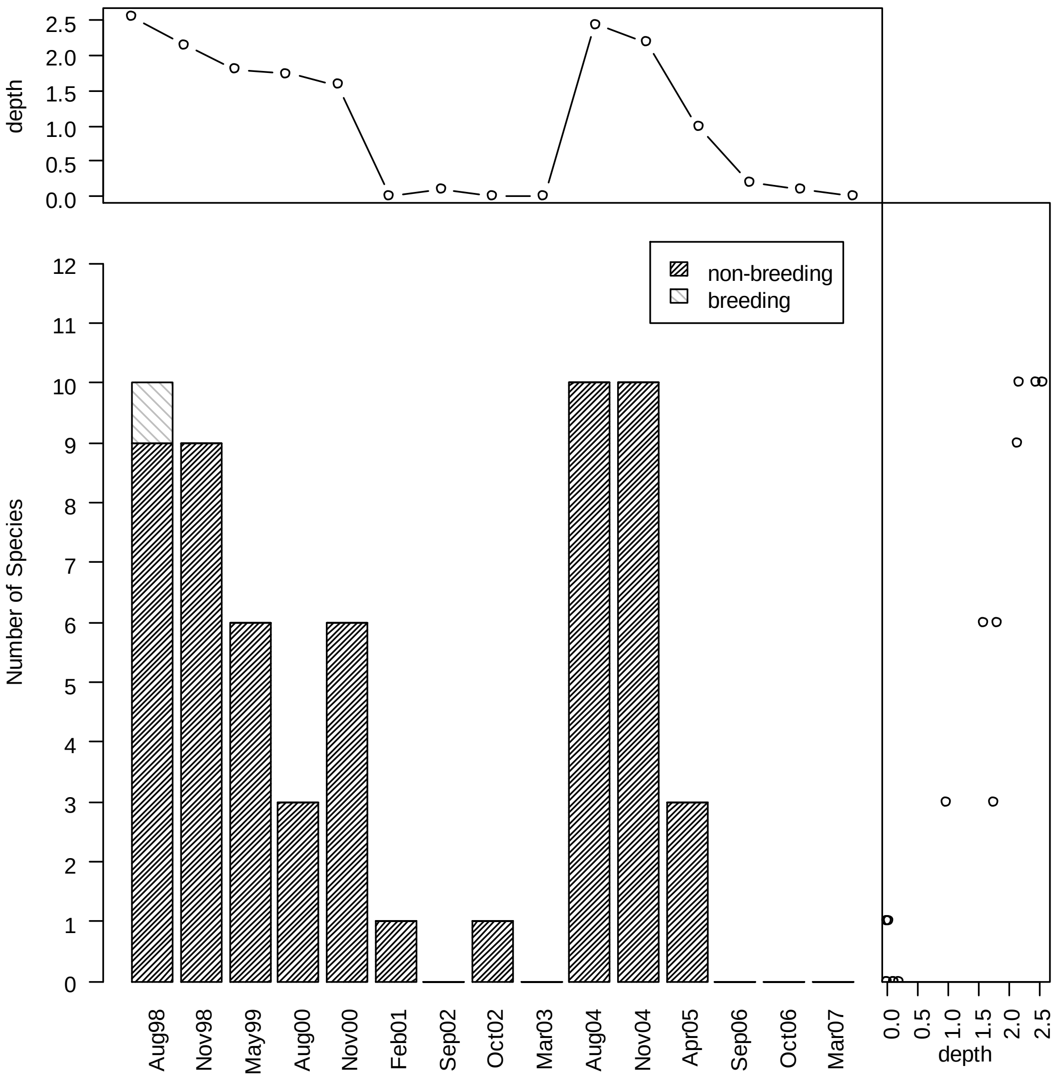


*Figure 5. Lake Eganu water bird species richness divided into breeding and non breeding species for each sampling occasion. Top panel is water depth by sampling occasion. Right panel shows total richness by water depth. Note discontinuous category (sample date) axis.*

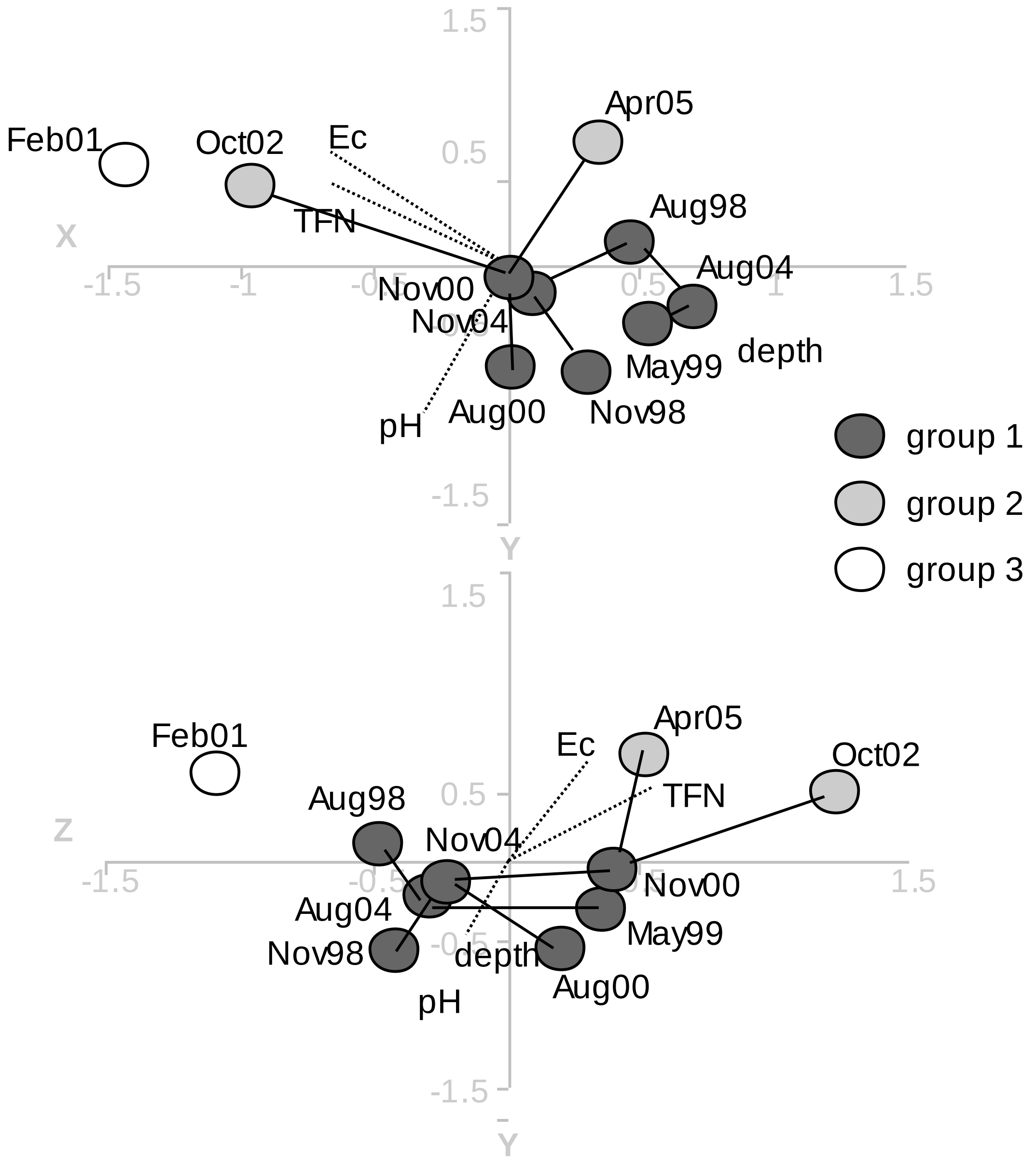


*Figure 6. Axes 1 and 2 [above] and axes 2 and 3, of a 3 dimension ordination (SSH) of Lake Eganu waterbird abundance, during individual surveys. Samples are linked by solid lines according to MST. PCC vectors with MCAO score<5% are displayed as dotted lines. Ordination stress was 0.11.*

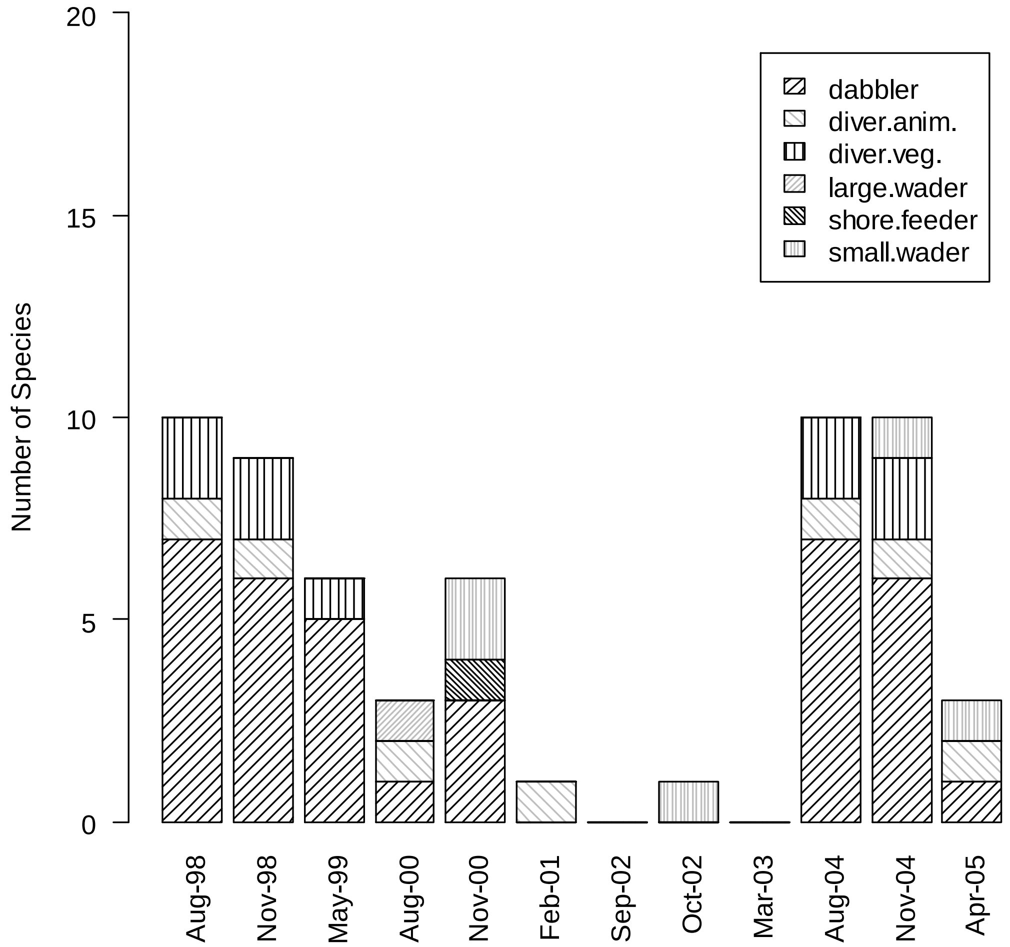


*Figure 7. Waterbird species richness distributed amongst feeding guilds for each sampling occasion at Lake Eganu.*

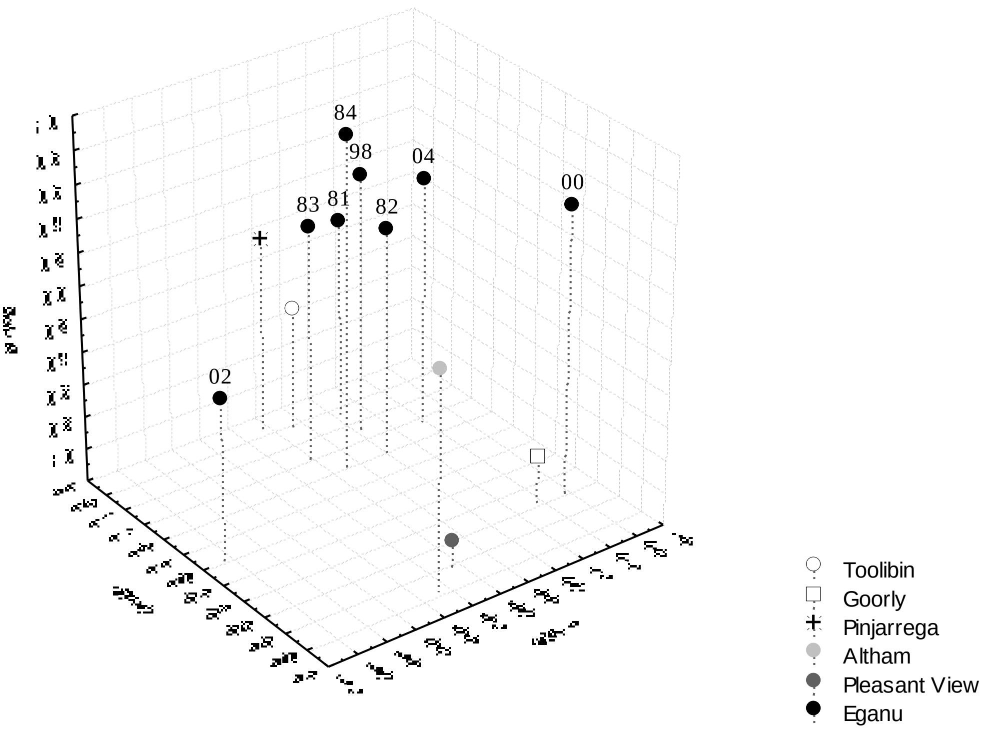


*Figure 8. Ordination (SSH Multi dimensional scaling) of Lake Eganu waterbird species presence/absence within a survey year, with five "marker" wetlands. Eganu data includes surveys from Jaensch et al. (1988). Ordination Stress = 0.11.*

## Invertebrates

*Description of invertebrate assemblage*

A total of 38 invertebrate taxa were collected from the wetland during the four monitoring years which spanned an eight year period (Table 6). Maximum species richness was 27 species in 1998. Species belonging to four of the invertebrate assemblages described by Pinder *et al.* (2004) for wheatbelt wetlands, were present at Lake Eganu during the monitoring period. Of these, assemblage H included the greatest number of species (16) and the highest proportion of species richness in all years, except 2002 when it was absent. Assemblage H is described as having an affinity for sub-saline (3-10 g/L) to saline (> 10 g/L) wetlands and occurs in secondarily salinised systems. Many of the species from this assemblage occurring at Lake Eganu were ostracods. Assemblage E, comprising ubiquitous and frequently collected taxa with a preference for fresh (< 3 g/L) to sub-saline (3-10 g/L) water was also commonly encountered (8 species). Assemblage G was represented by three species, *Coxiella* sp., *Merideicyclops baylii* and *Daphniopsis pusilla*. This assemblage is associated with primarily saline wetlands and was encountered in all years with all three species present in 2004. *Merideicyclops baylii* was collected on all sampling occasions. A total of 10 species did not belong to the assemblages of Pinder *et al.* (2004). All species are common and widespread in south-west Western Australia.

Invertebrate species richness falls within the bands of 'moderate condition' (Sim *et al.* 2008) in 1998 and 2004 when water depth was high (>2m) but was in the range of 'low condition' at low water levels. While family richness mirrored the trend of species richness this index was within the 'low condition' range on all sampling occasions.

*Table 5. Waterbird species list and abundances for Lake Eganu from three surveys each sampling year. % Occurrence is the proportion of surveys, with depth greater than 0 m, for which the species was recorded. The wetland was dry in March of the 2006 monitoring year. Figures in parentheses are rank score for the ten most frequently recorded species of Jaensch et al. (1988).*

| | **1998** | | | **2000** | | | **2002** | | | **2004** | | | **2006** | | **% Occ.** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Species** | **Aug** | **Nov** | **May** | **Aug** | **Nov** | **Feb** | **Sep** | **Oct** | **Mar** | **Aug** | **Nov** | **Apr** | **Sep** | **Oct** | |
| Australian Shelduck (2) | | 3224 | 10 | 193 | 387 | | | | | 2 | 323 | | | | 50.0 |
| Grey Teal (1) | 115 | 377 | 32 | | 3 | | | | | 83 | 197 | | | | 50.0 |
| Hoary-headed Grebe | 12 | 104 | | 6 | | | | | | 60 | 253 | 1 | | | 50.0 |
| Pink-eared Duck | 49 | 104 | 3 | | | | | | | 6 | 2 | 5 | | | 50.0 |
| Black Swan (5) | 3 | 297 | | | 7 | | | | | 7 | 84 | | | | 41.7 |
| Musk Duck (8) | 3 | 1 | 7 | | | | | | | 2 | 1 | | | | 41.7 |
| Pacific Black Duck (3) | 3 | 2 | 6 | | | | | | | 4 | 1 | | | | 41.7 |
| Australasian Shoveler | 17 | 72 | | | | | | | | 21 | 15 | | | | 33.3 |
| Eurasian Coot (6) | 200 | 1215 | | | | | | | | 50 | 260 | | | | 33.3 |
| Banded Stilt | | | | | 41 | | | 1 | | | | 938 | | | 25.0 |
| Chestnut Teal | 2 | | 1 | | | | | | | | | | | | 16.7 |
| Hardhead | 5 | | | | | | | | | 37 | | | | | 16.7 |
| Little Black Cormorant | | | | | | 1 | | | | | | | | | 8.3 |
| Red-capped Plover | | | | | 2 | | | | | | | | | | 8.3 |
| Red-necked Avocet | | | | | | | | | | | 3 | | | | 8.3 |
| Silver Gull (10) | | | | | 1 | | | | | | | | | | 8.3 |
| White-faced Heron (4) | | | | 8 | | | | | | | | | | | 8.3 |
| No Species | | | | | | | | | 0 | | | | 0 | 0 | 0.0 |
| Depth At Gauge* | 2.56 | 2.15 | 1.8 | 1.74 | 1.58 | 0 | 0.7 | 0.61 | 0 | 2.44 | 2.18 | 0.98 | 0.79 | 0.69 | |

* Depth gauge reads 60cm at the lake bed in the area surveyed so that non-zero values are inflated by 0.6m

*The invertebrate community over time*

During monitoring there was a negative correlation between species richness and salinity (EC) ($r = -0.987$; df = 3, $p<0.001$ see Fig. 9), but this is strongly influenced by very high salinities in 2002.

Ordination revealed as much difference between invertebrate communities from year to year as is observed between ‘marker’ wetlands (Fig. 10). In 1998 the community resembled that of Yaalup Lagoon which, while brackish, maintains a relatively high diversity of species. This reflects the high water levels and reduced salinity in lake Eganu at this time. In contrast, the invertebrate community in 2000, was closely aligned with Lake Parkeyerring, a strongly salinised wetland. The species poor community collected in 2002 is not reflective of the general condition of the lake, but rather reflects the specific highly saline conditions prevalent in the lakes final stages of drying.

The invertebrate community of 2004 is anomalous in that it does not appear similar to either ‘marker’ or other Lake Eganu samples despite occurring under similar conditions of EC, depth and pH to that present in 1998. While samples from 2004 and 1998 had similar species richness they were broadly separated in ordination space (Fig. 10) and shared only 16 species (of 37 species collectively). Comparison of invertebrate assemblages present (*sensu* Pinder *et al.* 2004) indicates a shift from dominance of assemblage H in 1998 to equivalence of assemblages H and E in 2004 (Fig. 11). These assemblages prefer sub-saline to saline (i.e. > 3 g/L) and fresh to sub-saline 9i.e. <10 g/L) conditions respectively, perhaps indicating that salinity had been lower in the months prior to sampling in 2004 compared to 1998. Invertebrate assemblage composition can reflect recent past conditions as well as current ones. Assemblage E species collected in 2004 included a high proportion of insects with dispersal capabilities (e.g. *Necterosoma penicillatus*, *Procladius paludicola* and *Austrolestes annulosus*).

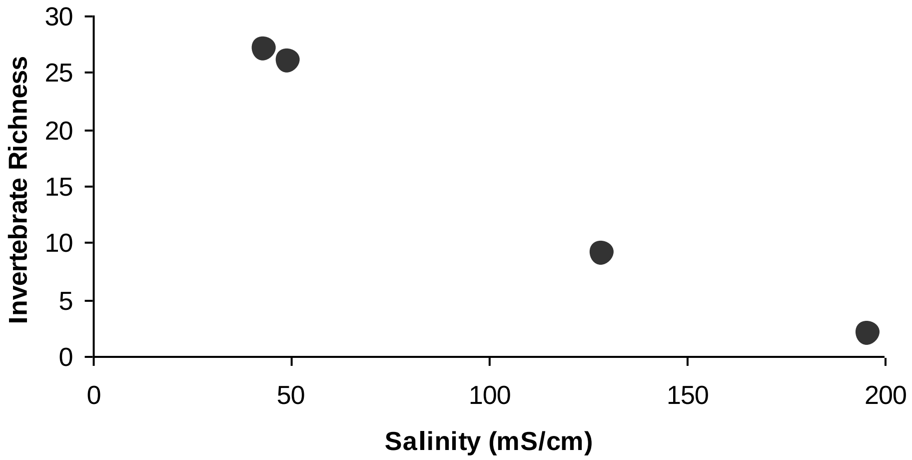


*Figure 9. Species richness of invertebrates against salinity (EC) at Lake Eganu.*

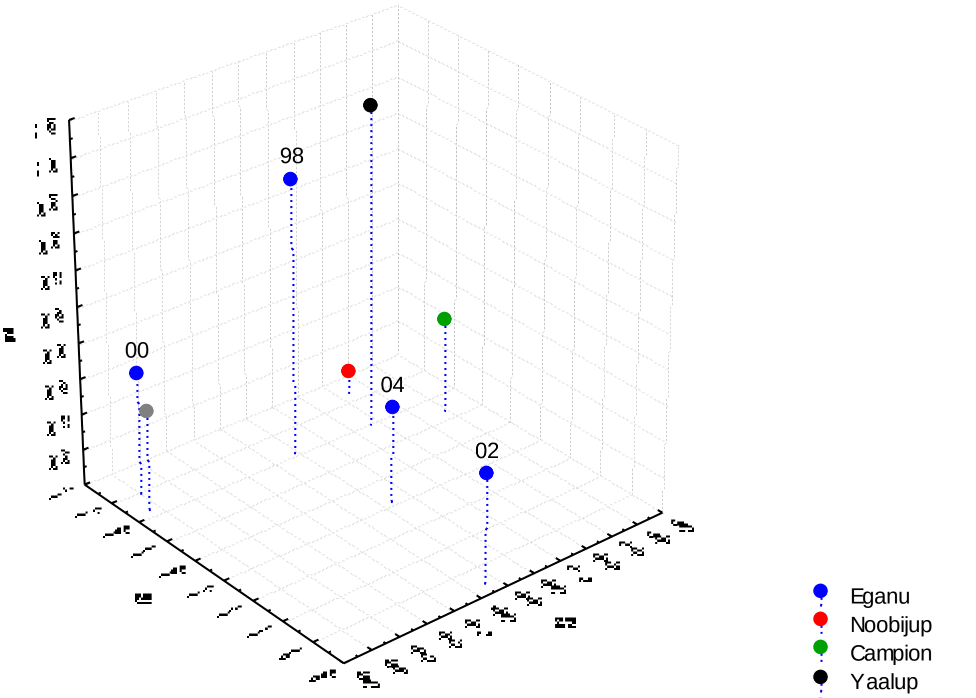


*Figure 10. SSH ordination of invertebrate species presence/absence at Lake Eganu and the four "marker' wetlands.*

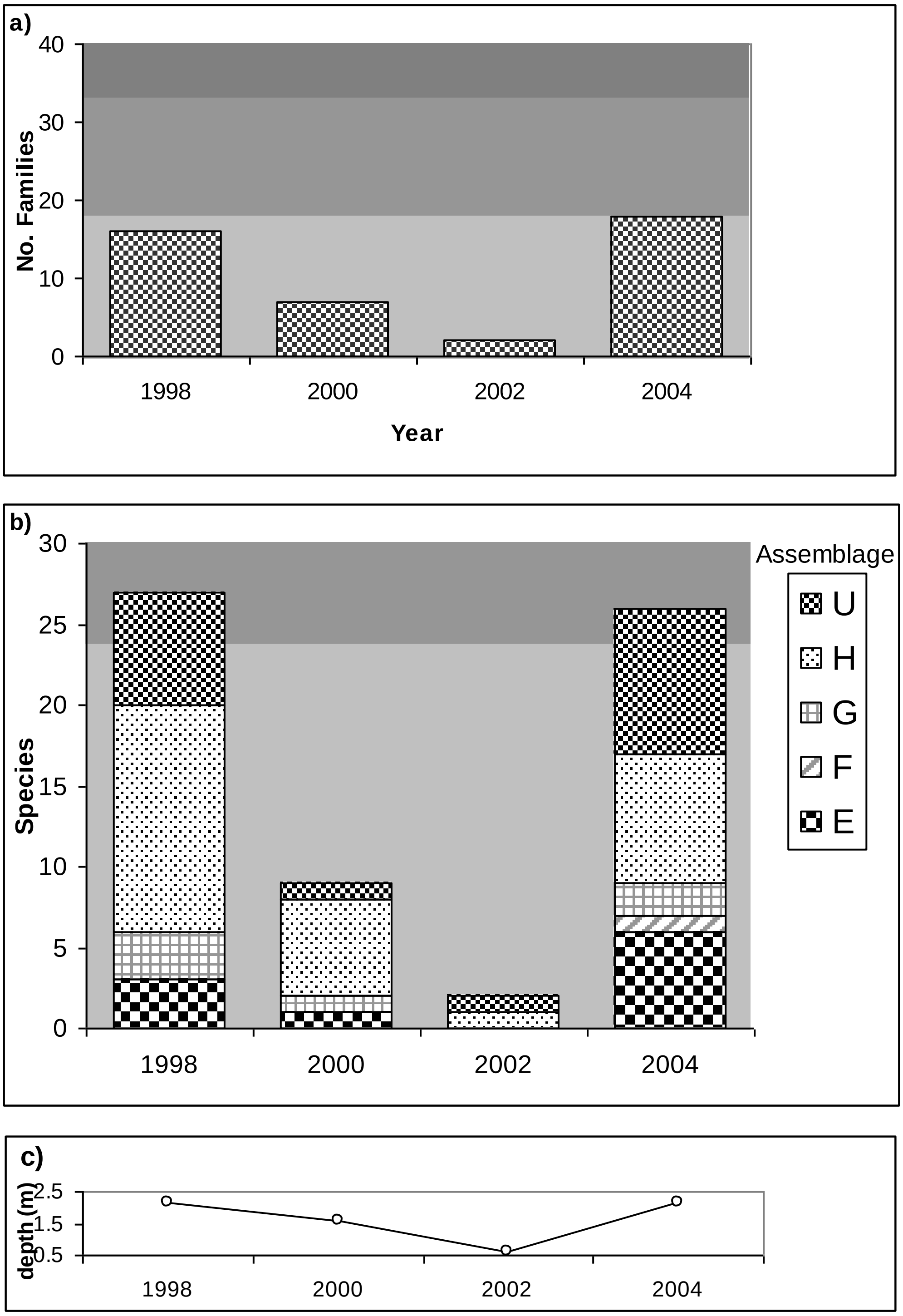


*Figure 11. Invertebrate species richness for the three sampling years at Lake Eganu. a) family richness, b) Richness of species within species assemblages of Pinder et al. (2004), c) depth at gauge during sampling. Background colour bands represent the "good" (dark grey), "medium" (mid grey) and "low" (light grey) condition reference ranges of Sim et al. (2008).*

*Table 6. Lake Eganu invertebrate species list. Assemblages are from Pinder et al. (2004).*

| **Lowest Id** | **National Code** | **1998** | **2000** | **2002** | **2004** | **Assemblage** |
|---|---|---|---|---|---|---|
| Turbellaria | IF999999 | 2 | | | | |
| Nematoda | II999999 | 2 | | 1 | | |
| ROTIFERA | | | | | | |
| *Hexarthra fennica* | JF040105 | 2 | | | | H |
| GASTROPODA | | | | | | |
| *Coxiella* sp. | KG130299 | 2 | 2 | | 2 | G |
| OLIGOCHAETA | | | | | | |
| Naididae | LO059999 | | | | 1 | |
| ANOSTRACA | | | | | | |
| *Parartemia* sp. | OD020199 | | | | 2 | |
| CLADOCERA | | | | | | |
| *Daphniopsis pusilla* | OG040301 | | | | 2 | G |
| *Daphniopsis queenslandensis* | OG040302 | 2 | | | | H |
| *Daphniopsis truncata* | OG040305 | 2 | | | | H |
| OSTRACODA | | | | | | |
| *Australocypris insularis* | OH080203 | 2 | 2 | | 2 | H |
| *Cyprinotus edwardi* | OH080602 | 2 | | | 2 | H |
| *Diacypris dictyote* | OH080701 | 2 | | | 2 | H |
| *Diacypris spinosa* | OH080703 | 1 | | | | H |
| *Diacypris compacta* | OH080704 | 1 | 2 | | 2 | H |
| *Mytilocypris ambiguosa* | OH081201 | | | | 1 | E |
| *Mytilocypris mytiloides* | OH081204 | 2 | | | 2 | H |
| *Reticypris clava* | OH081501 | 2 | | | | H |
| *Platycypris baueri* | OH082601 | 2 | 2 | | 2 | H |
| COPEPODA | | | | | | |
| *Calamoecia clitellata* | OJ110208 | 2 | 2 | | 2 | H |
| *Apocyclops dengizicus* | OJ311201 | 2 | | | | H |
| *Meridiecyclops baylyi* | OJ311701 | 2 | 1 | 1 | 2 | G |
| *Mesochra* nr *flava* | OJ6103A1 | 2 | | | 2 | H |
| AMPHIPODA | | | | | | |
| *Austrochiltonia subtenuis* | OP020102 | | | | 1 | E |
| ISOPODA | | | | | | |
| *Haloniscus searlei* | OR250101 | 2 | 2 | | | H |
| COLEOPTERA | | | | | | |
| *Necterosoma penicillatus* | QC092001 | | 2 | | 2 | E |
| *Berosus* sp. (larvae) | QC110499 | 2 | | | 2 | |
| DIPTERA | | | | | | |
| *Monohelea* sp. 1 (SAP) | QD0919A0 | | | | 2 | E |
| Stratiomyidae | QD249999 | 2 | | | 2 | |
| Dolichopodidae | QD369999 | 1 | | | 2 | |
| Ephydridae sp. 3 (SAP) | QD7899A7 | 1 | | | | |
| Ephydridae sp. 7(SAP) | QD7899B1 | | | | 1 | |
| Muscidae sp. A (SAP) | QD8999A0 | | | | 1 | |
| *Procladius paludicola* | QDAE0803 | 2 | | | 2 | E |
| *Tanytarsus barbitarsis* | QDAH0402 | | 2 | | | H |
| *Cladopelma curtivalva* | QDAI2201 | 2 | | | 1 | E |
| ODONATA | | | | | | |
| *Austrolestes analis* | QO050101 | 1 | | | | F |
| *Austrolestes annulosus* | QO050102 | | | | 2 | E |
| *Austrolestes io* | QO050105 | 1 | | | 1 | E |
| | **Richness** | **27** | **9** | **2** | **26** | |

## Vegetation

*Transect inundation*

Coupling of transect elevation and wetland depth data was used by Lyons *et al.* (2007) to reconstruct the inundation history of transects during wetland fill events. Inundation of transect 2 was determined to have been complete on three occasions (1983, 1993 and 1999), with partial inundation of the low lying swale occurring on a further 11 occasions since depth data records commenced in 1979. Transect 1 is likely to have been partially inundated in 1999 (see figure 8(a), Lyons *et al.* 2007). Wetlands depths since 2004 (the end of the Lyons *et al.* 2007 study) have not inundated transects.

*Shallow groundwater*

Elevated groundwater levels at the beginning of the monitoring period probably resulted from the high rainfall (relative to subsequent years) and lake fill events of 1998 and 1999 (Fig. 12). At Lake Eganu and the subsidiary wetland sampled by transect 3, groundwater levels declined from 1m to almost 2m below the surface from 1999 to 2002, with little apparent seasonal variation. This decline corresponded to low rainfall and concomitant low water levels in the lake (Fig. 3). Higher rainfall during 2003 and 2004 was associated with the reestablishment of a marked seasonal pattern of rise and fall of groundwater, and during spring 2004 groundwater approached the surface at transects 1 and 3 (Fig. 12). From 2005 the seasonal amplitude declined and depth to groundwater increased, with several bores becoming dry (Fig. 12).

Over the same period groundwater salinity declined from the commencement of monitoring in 1999 to 2003/2004 (decline from ca. 6400 mS/m to 1200mS/m) when higher rainfall was reflected in seasonal recharge and dilution (Fig. 13). Groundwater salinity was higher in the spring of 2005 and 2006 (ca. 5500 mS/m) associated with lower rainfall although interpretation is difficult given the lack of data from dry bores. Throughout the monitoring period transect 2 consistently showed the highest groundwater salinity (exceeding 6000 $mSm^{-1}$ on three occasions) with groundwater also being within 1 m of the surface on four occasions since 1999 (Fig. 13).

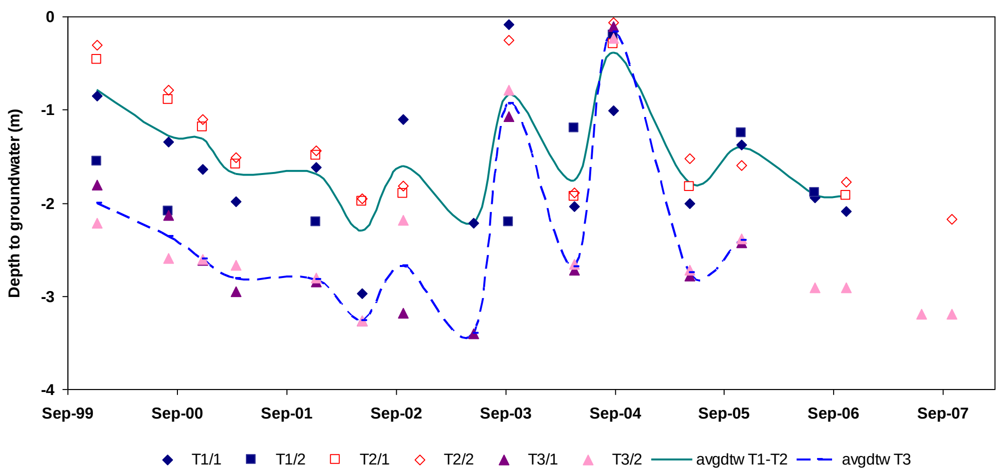


*Figure 12. Depth to groundwater at Lake Eganu. Separate trendlines (means) for depths to groundwater are shown for transects 1 & 2 combined and transect 3.*

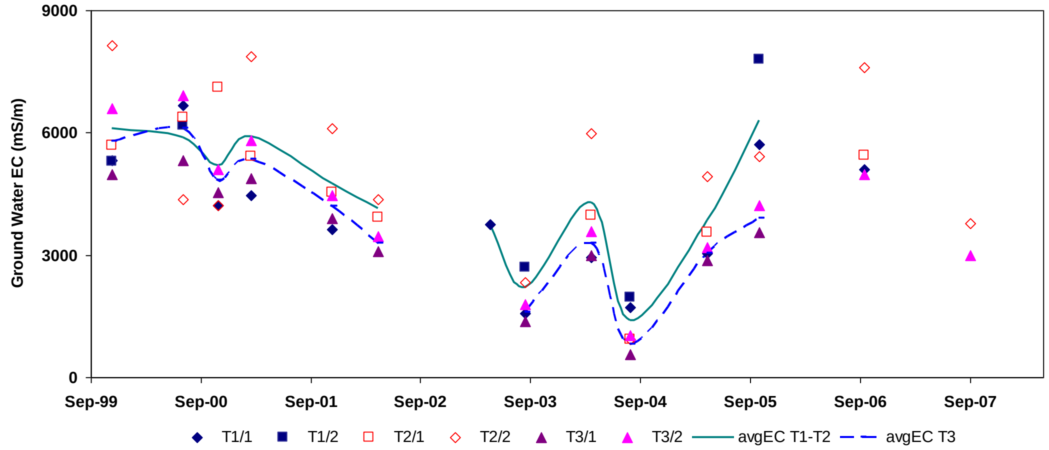


*Figure 13. Groundwater electrical conductivity at Lake Eganu. Separate trendlines (means) for groundwater conductivity are shown for transects 1 & 2 combined and transect 3.*

## Soil conductivity

Patterns of soil conductivity ($EC_a$) at transects 1 and 2 were strongly related to the elevation profiles of each transect. At both transects maximum $EC_a$ values, for both horizontal and vertical instrument orientations, were recorded within the shallow swale between 15 – 30m along the transects, and at the wetland end of transect 2 (Fig. 14, 15). At transect 1, mean $EC_a$ (vertical dipole) exceeded 350 mS $m^{-1}$ within the swale in all years. At transect 2, mean surface conductivity within the swale exceeded 400 mS $m^{-1}$ at all times except 2007. At the wetland edge of transect 2 (50 – 60 m) conductivity values exceeded 600 mS m-1 in 1998. This part of transect 2 represented the upper edge of the wetland basin and sampled the highly saline clays of the lake shore. Lowest conductivity values were recorded on higher elevations associated with the upland end of transects and low sandy rises closer to the wetland (Fig. 15).

Over the monitoring period the spatial pattern and magnitude of $EC_a$ values for transect 1 appears relatively stable with the exception of an increase in conductivity within the swale in 2004 (Fig. 14). This increase may reflect the proximity of shallow saline groundwater to the surface at this time (see Fig. 12). At transect 2 while the spatial pattern remained stable from 1998 to 2007, a modest decline in conductivity is evident (Fig. 15).

Taken across the entire transect the lowest $EC_a$ values were recorded at transect 3, where conductivity has declined over the monitoring period with a significant drop between 1998 and 2001 (Fig. 16).

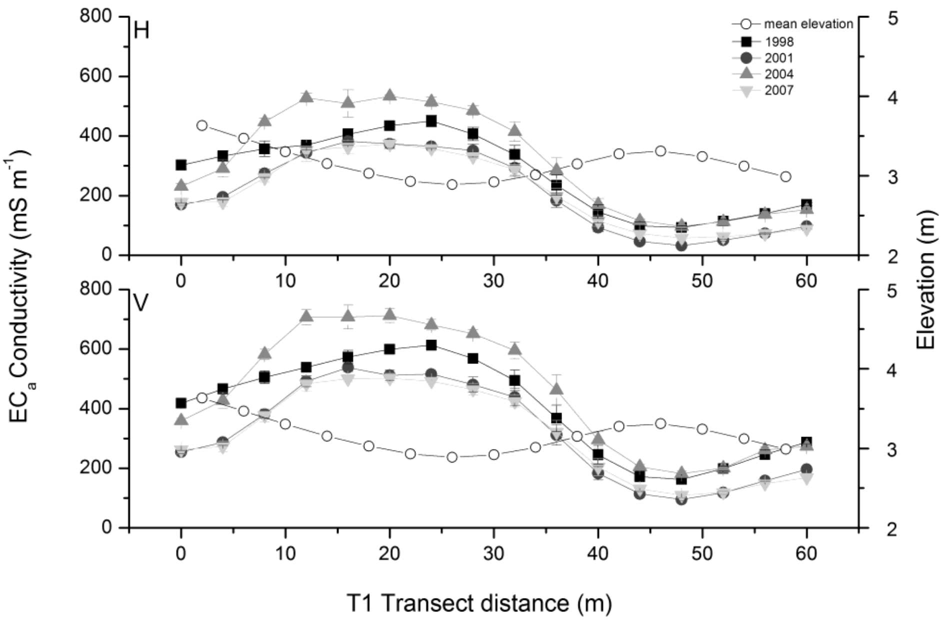


*Figure 14. Soil conductivity ($EC_a$ ) along transect 1 for Lake Eganu covering the period 1998 to 2007. $EC_a$ values are the mean of three points at both sides and the centre of each distance along the transect (error bars +/- 1 SD). The end closest to the lake is at 60m. EM38 measurements in horizontal dipole (H), vertical dipole (V) orientations.*

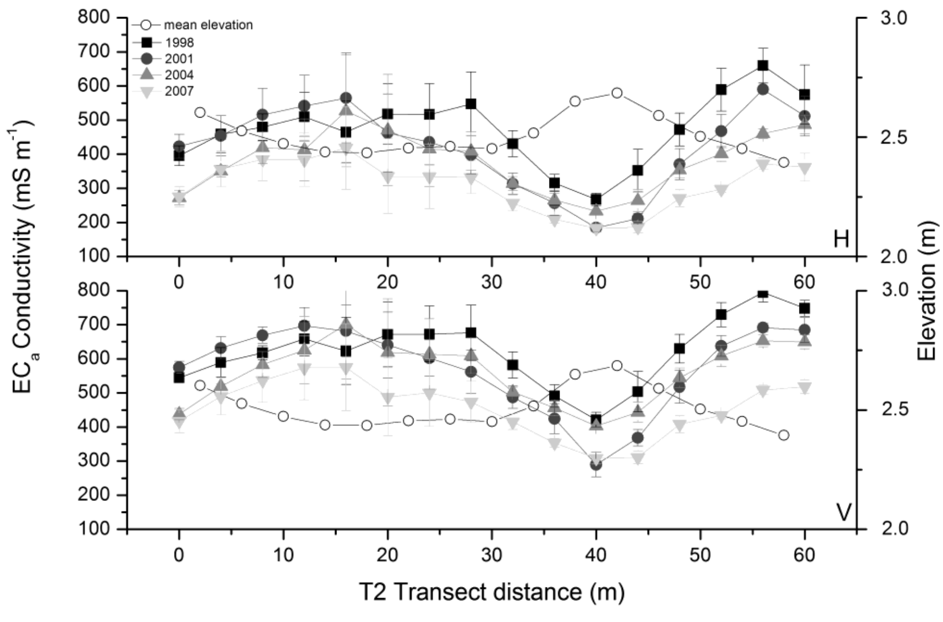


*Figure 15. Soil conductivity ($EC_a$) along transect 2 for Lake Eganu covering the period 1998 to 2007. $EC_a$ values are the mean of three points at both sides and the centre of each distance along the transect (error bars +/- 1 SD). The end closest to the lake is at 60m. EM38 measurements in horizontal dipole (H), vertical dipole (V) orientations.*

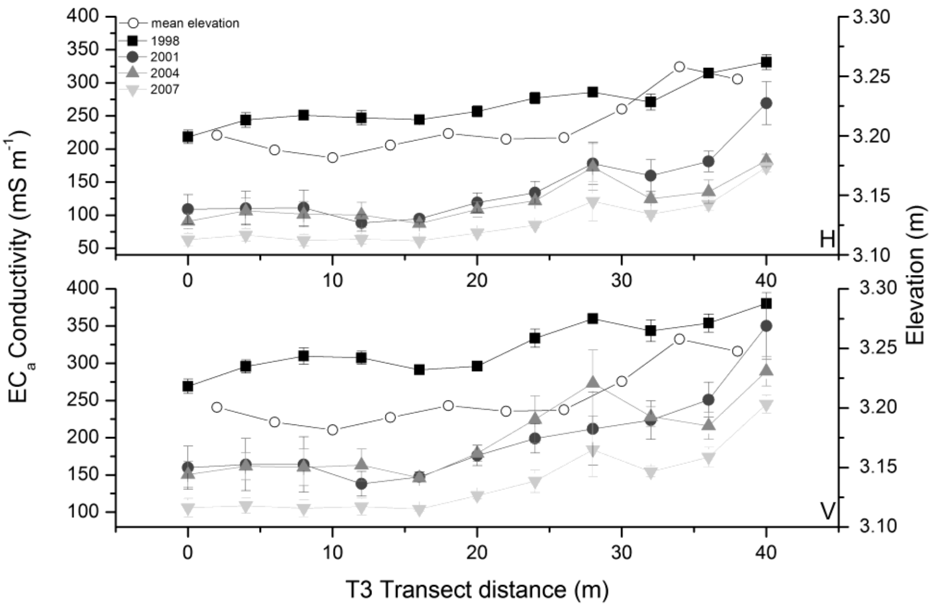


*Figure 16. Soil conductivity (EC$_a$) at transect 3 for Lake Eganu covering the period 1998 to 2007. EC$_a$ values are the mean of three points at both sides and the centre of each distance along the transect (error bars +/- 1 SD). The end closest to the lake is at 60m. EM38 measurements in horizontal dipole (H), vertical dipole (V) orientations.*

## Vegetation trends

*Trends in basal area and population structure*

*Casuarina obesa* dominated both transects on the margin of Lake Eganu (transects 1 and 2), and was a codominant with *M. strobophylla* at the subsiduary wetland sampled by transect 3. A small stand of *Eucalyptus loxophleba* (3 individuals) occurred at the upland end of transect 1 and persisted over the study period.

Within transect 1, total basal area for *C. obesa* declined in the low lying swale area between 20m and 30m along the transect (Fig. 17a). For the monitoring period 1998 – 2007 total basal area loss due to tree death was 22.7% with the decline occurring primarily after 2001.

The *C. obesa* stand at transect 1 included a large band of much earlier recruitment (predating the commencement of monitoring) with most trees being less than 150 mm DBH. A small number of seedlings (<20 mm size class) were recorded in all monitoring years but overall there was no net recruitment. Primarily there was a decline in the number of individuals within size classes <200 mm DBH, with some larger trees persisting (Fig. 17b).

Transect 2 was a pure stand of *C. obesa* sampling a lower elevation than transect 1 (2.4 m to 2.7 m *cf.* 2.9 m to 3.6 m) (Fig. 17a, 18a). Total basal area for *C. obesa* declined across most of the transect over the study period with a total loss of 77.2% from 1998 to 2007. All trees at elevations less than 2.5 m died, with this largely occurring between 2001 and 2004 (individual year data not shown). Where the elevation was slightly higher (a small sandy rise) survival and some growth of a limited number of trees occurred (Fig. 18a). Death of individuals occurred across most size classes and no seedlings were recorded during the study period (Fig. 18b).

At transect 3 both *M. strobophylla* and *C. obesa* accumulated basal area over the full monitoring period from 1999 – 2007 (7.8% and 10.8% increase respectively). For *C. obesa* this trend was relatively uniform across the transect (Fig 19a). The pattern for *M. strobophylla* was more variable with most areas showing some increase, but the far end of the transect (36m – 40m) showing modest decline associated with limited tree death (Fig. 20a). A separate wetland adjacent to transect 3 has seen the total collapse of mature *M. strobophylla* stands over the study period. The size class distribution reflects the growth of individuals with some movement into larger size classes evident. Very few seedlings were recorded and no individuals entered the "adult" population (Fig. 20b).

Examination of basal area data for *C. obesa* pooled across transects reveals a strong negative relationship between basal area and sub-plot $EC_a$ (vertical dipole) (Fig. 21). For $EC_a$ greater than ca. 450 mS $m^{-1}$ most subplots declined in basal area. EM38 data for the shallower part of the soil profile ($EC_a$ horizontal dipole) showed the same pattern against basal area change with the conductivity values consistently lower by ca. 100 mS $m^{-1}$ across most sub-plots. Two subplots accumulated basal area despite mean $EC_a$ (vertical dipole) exceeding 450 mS $m^{-1}$. Both subplots occurred at the change in slope at the edge of sandy rises.

Basal area data (transect 1 and 2) were also examined against sub-plot elevation. Each transect showed a separate positive relationship between elevation and accumulation in basal area (Fig. 22). These data also reveal that while the lowest elevation transect (T2) has seen the greatest decline in basal area (ca. 77% vs. 22%), decline is not explained simply by sub-plot elevation relative to the wetland (ostensibly in response to inundation) but also dependent on subplot soil attributes. and small scale topography at the transect scale. Relative elevation at the transect scale is in part a measure of the depth of loamy sand overlaying the sandy clay base of the broader wetland floor.

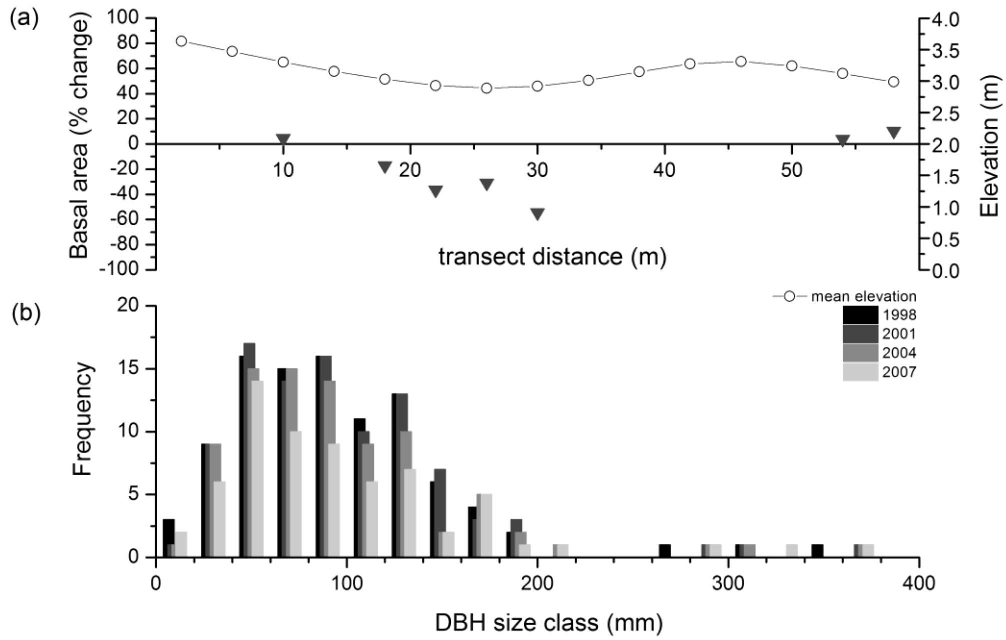


*Figure 17. Casuarina obesa at transect 1. **(a)** Change in stand basal area (1998 -2007) for each 4 x 20m subplot along the transect (solid triangles). Higher transect distance is closer to the wetland. **(b)** Frequency histogram of diameter at breast height (DBH) for all sample years.*

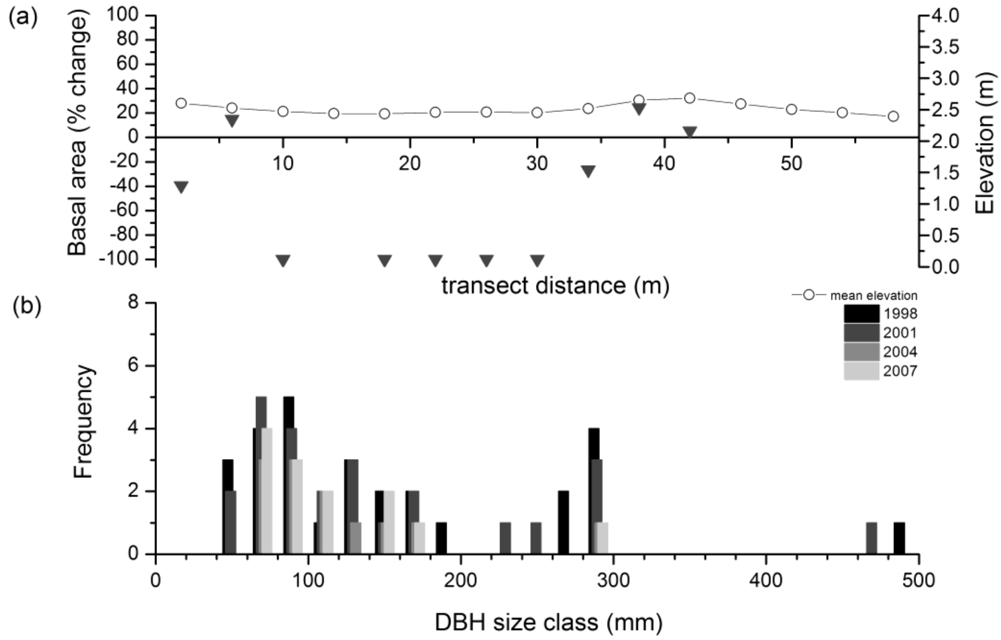


*Figure 18. Casuarina obesa at transect 2. **(a)** Change in stand basal area (1998 -2007) for each 4 x 20m subplot along the transect (solid triangles). Higher transect distance is closer to the wetland. **(b)** Frequency histogram of diameter at breast height (DBH) for all sample years.*

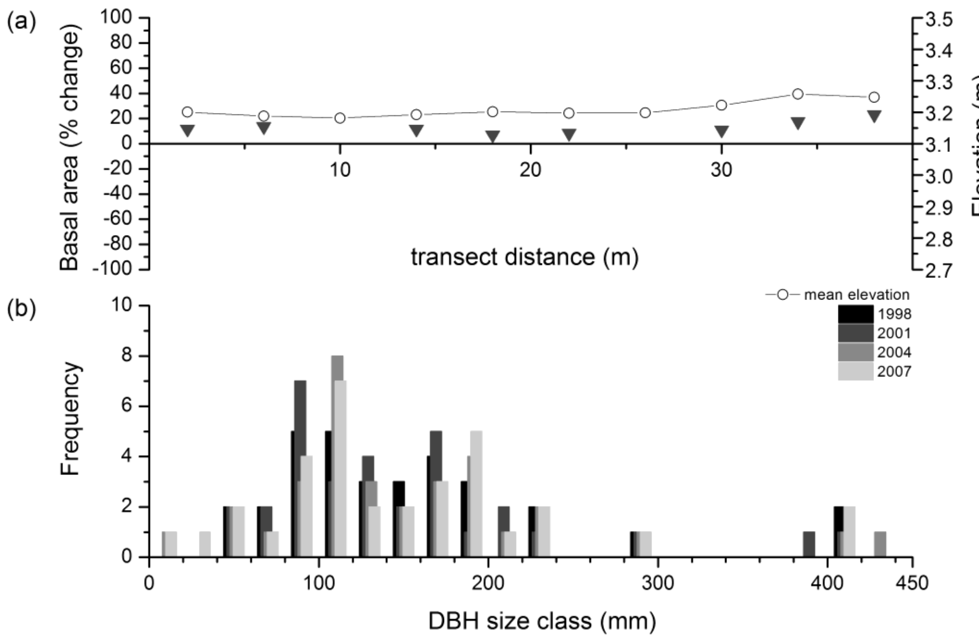


*Figure 19. Casuarina obesa at transect 3. **(a)** Change in stand basal area (1998 -2007) for each 4 x 20m subplot along the transect (solid triangles). **(b)** Frequency histogram of diameter at breast height (DBH) for all sample years.*

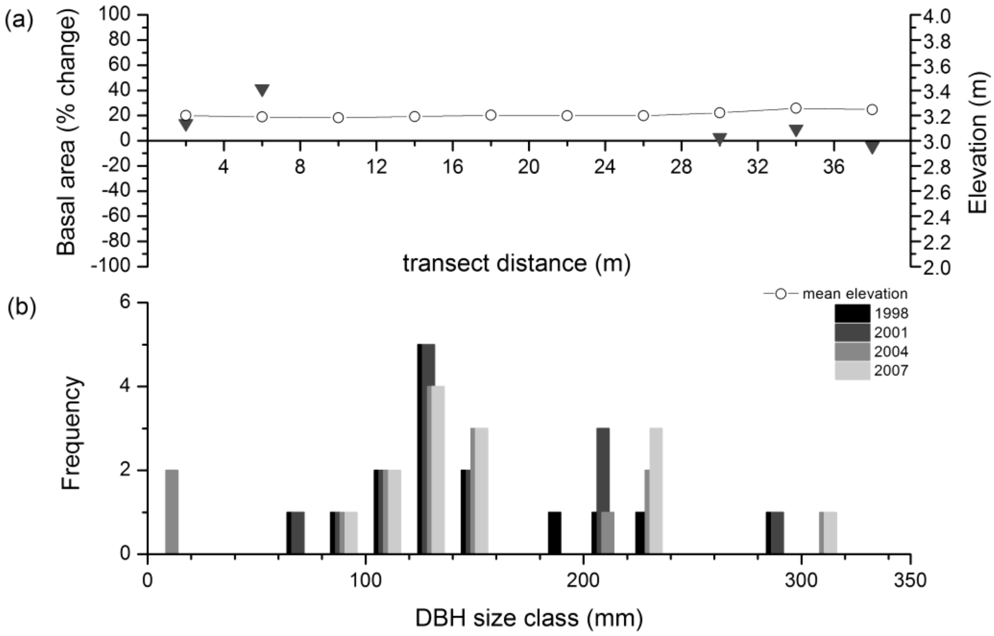


*Figure 20. Melaleuca strobophylla at transect 3. **(a)** Change in stand basal area (1998 -2007) for each 4 x 20m subplot along the transect (solid triangles). **(b)** Frequency histogram of diameter at breast height (DBH) for all sample years. 2007.*

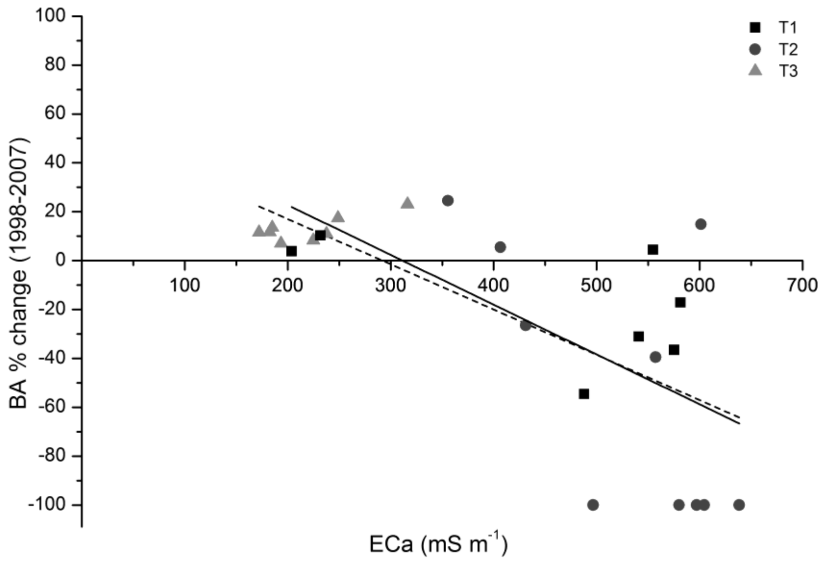


*Figure 21. Percentage change in sub-plot basal area (BA) for Casuarina obesa plotted against sub-plot soil electrical conductivity ( EM38 – $EC_a$ – vertical dipole) (average of all years data) for transects at Lake Eganu and the subsidiary wetland (transect 3). Linear regressions are shown for all transects pooled (dashed line – $R^2$ = 0.47259, p = <0.001) and for transects 1 and 2 at Lake Eganu pooled (solid line- $R^2$ = 0.28025, p = <0.05).*

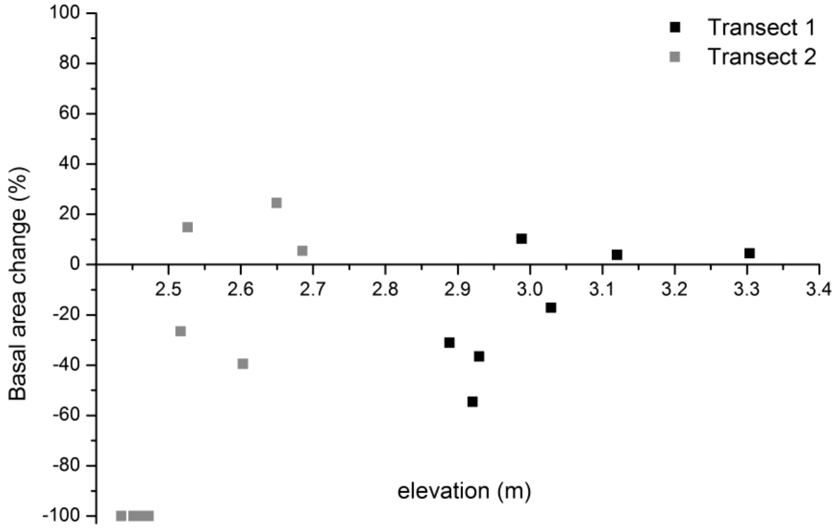


*Figure 22. Percentage change in sub-plot basal area for Casuarina obesa at Lake Eganu, plotted against sub-plot mean elevation.*

*Trends in canopy condition*

Transect 2 showed significant differences between years and a significant decline in the condition of *C. obesa* for both the stand and extant trees, between 1998 and 2007 (Figure 23a, b; Table 7). A similar result for the *C. obesa* stand was seen at transect 1, yet surviving trees showed no significant difference in condition between 1998 and 2007 (Figure 23a, b; Table 7). At transect 3 *M. strobophylla* condition declined between 1998 and 2007 yet the condition of *C. obesa* was not significantly different over the monitoring period (Figure 23a, b; Table 7).

The condition of extant of *C. obesa* appears similar at transect 1 and 2 for all sampling years and reflects the stressed nature of surviving trees within the riparian zone of the lake. In contrast *C. obesa* at the upstream wetland remains relatively healthy.

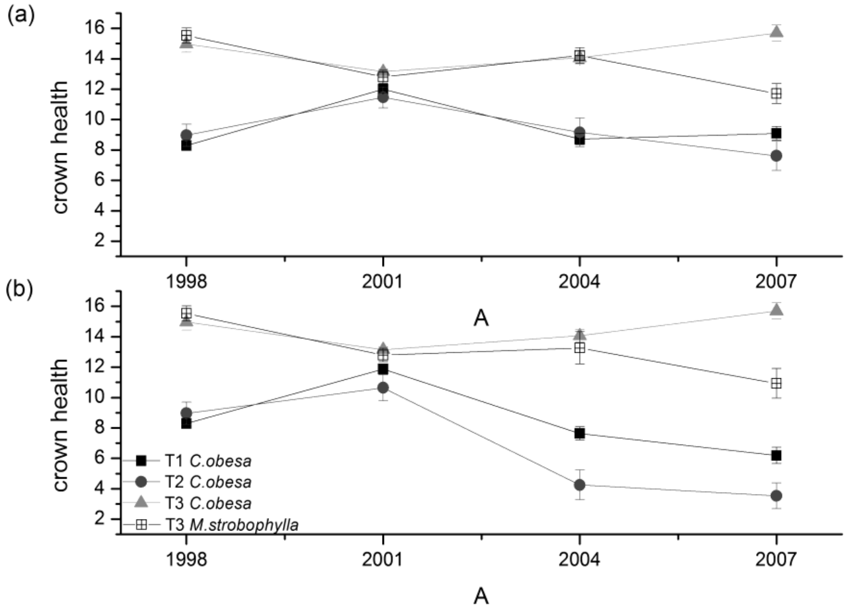


*Figure 23. Casuarina obesa mean (+/- 1 SE) aggregate crown health scores (3 – extremely stressed, 23 - max health) for transects 1-3.* ***(a)*** *Means of trees extant at each sample time.* ***(b)*** *Means calculated by retaining trees that died after the first and subsequent sample years by assigning a zero condition score.*

*Table 7. Chi-square values for Friedman non-parametric ANOVA of tree health ratings over sample times (1998. 2001, 2004, 2007), and Wilcoxon paired test comparing 1998 data to 2007 condition ratings. The first in each couplet of test results includes zero scores for dead trees. The second test statistic only includes individual that were alive at all samples times tested. A single result implies all trees survived the monitoring period.*

| | **Transect 1** | **Transect 2** | **Transect 3** |
|---|---|---|---|
| **Freidman ANOVA χ-square (N, df)** | | | |
| *Casuarina obesa* | 128.33 (95, 3) ++ | 51.24 (28, 3) ++ | 30.55 (32, 3) NS |
| | 80.83 (65, 3) ++ | 23.54 (13, 3) p ++ | |
| *Melaleuca strobophylla* | - | - | 24.05 (15,3) ++ |
| | | | 23.26 (14, 3) **** |
| **Wilcoxon paired test Z (N)** | | | |
| *Casuarina obesa* | 3.38 (95) **** | 4.24 (28) **** | 1.52 (32) NS |
| | 1.78 (65) NS | 2.29 (13) * | |
| *Melaleuca strobophylla* | - | - | 2.93 (15) *** |
| | | | 2.80 (14) ** |

NS, *P*> 0.05, *, *P* <0.05, **, P<0.01, ***, P <0.005, ****, P <0.0005, ++, P<0.00001

## DISCUSSION

### Biodiversity components of the water body

The secondary salinisation of Lake Eganu has had substantial impacts on the biodiversity of this wetland since the 1960s. Lyons *et al.* (2007) have suggested that up to 80% of the original invertebrate fauna may have been lost. There is also evidence that bird communities have changed, with at least eleven species recorded during the 1980s no longer occurring (Lyons *et al.* 2007 and data reported here). While directly influenced by the increased salinity of Lake Eganu, the absence of these species is also likely to be linked to the loss of the fringing band of living trees that encompassed the lake's margin until the 1970s (Lyons *et al*. 2007). The loss of vegetation has undoubtedly also had a major impact on the value of the wetland for breeding waterbirds. Jaensch *et al.* (1988) recorded breeding amongst 8 species during the 1980s. Seven of these were recorded during the current monitoring program but only one (Grey Teal) was observed breeding. The waterbird fauna is now dominated by species that are generally common and ubiquitous across the Wheatbelt because of their broad tolerances (e.g. to salinity, habitat type, land use etc.). The dominance of ubiquitous waterbird species is paralleled in the invertebrate community which is dominated by widespread salt-tolerant species.

Despite the continued influence of salinisation on all components of water body biodiversity (Fig. 24), when water levels are high and salinity low, the lake supports a significant abundance and diversity of waterbirds and the richness of invertebrates falls in the 'moderate' range for condition (*cf.* Sim *et al.* 2008). Up to 5000 birds may be present (November 1998) when not constrained by low depth and high salinity. These higher counts match those of earlier decades. Halse *et al.* (1992, 1994) counted 3570 waterbirds (from a restricted list of 13 species) in March 1989, 3600 in November 1990 and 5502 in March 1991 placing the wetland in the top 15 for abundance in a south-west regional count of waterfowl.

Lake Eganu is an important refuge for large numbers of moulting Australian Shelduck. These birds undergo a complete moult following the breeding season i.e. moulting in late spring through summer, and are flightless for a period of 26 days during the moult (Riggert 1977). The relative isolation of Lake Eganu from roads and human activities increases the importance of the lake to moulting Shelduck which are susceptible to predation and disturbance while flightless.

The suitability of a wetland for moulting Shelduck is also likely to be influenced by food reserves since the complete moult takes 150 days and the aggregations of birds that occur will consume large quantities of food. Australian Shelduck eat plant material such as *Lamprothamnium* sp and *Ruppia* sp. and aquatic invertebrates including ostracods, *Coxiella* sp., brine shrimp and dipteran larvae (Riggert 1969, Delroy 1973). All of these potential food sources are present at Lake Eganu, but show declining abundance at high salinities, such as occur at depths less than 1.5 m at the gauge. Australian Shelduck have a high degree of salinity tolerance and the primary factor affecting their presence at Lake Eganu for moulting are salinity conditions conducive for the growth of submerged macrophytes such as *Lamprothamnium*. Sim *et al*. (2006) report germination threshold salinities in the range 30-50 g/L for *Lamprothamnium cf succinctum*, *L. macropogon* and *Ruppia megacarpa*. Salinities above this threshold were observed in late winter 2000 and 2002 and are likely to have prevented the seasonal establishment of submerged macrophytes. Increased salinity in consecutive seasons may result in a regime shift such as described by Davis *et al.* (2003) in which submerged macrophytes may die out and the system changes from clear and macrophyte dominated to turbid and phytoplankton or microbial mat dominated. Such an altered system is unlikely to support either waterbird or invertebrate communities as presently encountered.

The continued persistence of moderately diverse invertebrate communities is dependant on (surface and groundwater) inflow salinities remaining at their current levels and sufficient rainfall occurring to periodically fill the wetland. While there may be some exporting of salt when the lake overflows, e.g. August 1998 and 2004, the salinity/depth relationship has remained constant over the monitoring period, suggesting that salt input and output are currently in balance. Any imbalance resulting in increased salinity is likely to result in reduced species richness, particularly with a gradual loss of species from assemblage E, as is currently observed at low lake depths.

A shift in invertebrate community composition between 1998 and 2004, despite similar conditions of salinity, suggests a different trajectory for community development in these two years. In 1998 there was a high proportion of taxa from assemblage H which show a preference for sub-saline to saline conditions (Pinder *et al*. 2004) and which, because of a low dispersal capability are dependent on propagules from within the lake to re-establish populations after refilling. By contrast, in 2004 there was a high proportion of assemblage E taxa particularly insects with dispersal capabilities (e.g. *Necterosoma penicillatus*, *Procladius paludicola* and *Austrolestes annulosus*). Given that these taxa are unable to persist through periods of high salinity they must re-establish from source populations in other wetlands. It is likely that in 2004 both a longer wetted period at Lake Eganu (keeping salinities lower for longer prior to sampling) and a widespread filling of other wetlands in the region facilitated greater colonisation by these dispersive taxa.

*Austrochiltonia subtenuis*, a member of assemblage E, does not have an inherent dispersal capability but may be dispersed by waterbirds. This species was collected in 1979 (Halse 1981) and its absence from 1998 samples led Cale *et al.* (2004) to speculate that Lake Eganu had become isolated from source populations that could recolonise the wetland. The presence of this species (and the many other assemblage E species) in 2004 indicates that Lake Eganu is not spatially isolated in this manner but may be dependent on dispersal vectors such as waterbirds for the persistence of some invertebrate species. The presence of a large number of species with dispersal traits and only moderate salinity tolerance (i.e. Assemblage E) in 2004 reinforces the view that the overall biodiversity of Lake Eganu is dependent on the condition of nearby wetlands which can act as source populations for these taxa which are unable to persist permanently at Lake Eganu.

The inter relationship of biodiversity, water depth and salinity observed at Lake Eganu is typical of secondarily salinised wetlands which still experience a substantial reduction in salinity when filled. Using cluster analysis of waterbird presence/absence data, Halse *et al* (1993) placed Lake Eganu with a group of 37 wetlands including Parkeyerring, Nonalling, Mears, Gore, Little White, Walbyring, Walyormouring and Ninan. A similar classification by Pinder *et al*. (2004) using waterbird, invertebrate and plants placed Eganu in a group including these same wetlands (and 16 others). It is likely that the biodiversity of these wetlands will share similar responses to environmental change. Where these lakes are unable to fill because of reduced rainfall or where inflows become more saline, reducing the dilution effect, biodiversity will show further degradation (Fig. 24). Conversely, where management aims to restore biodiversity in this wetland type, only small gains can be expected without addressing these issues.

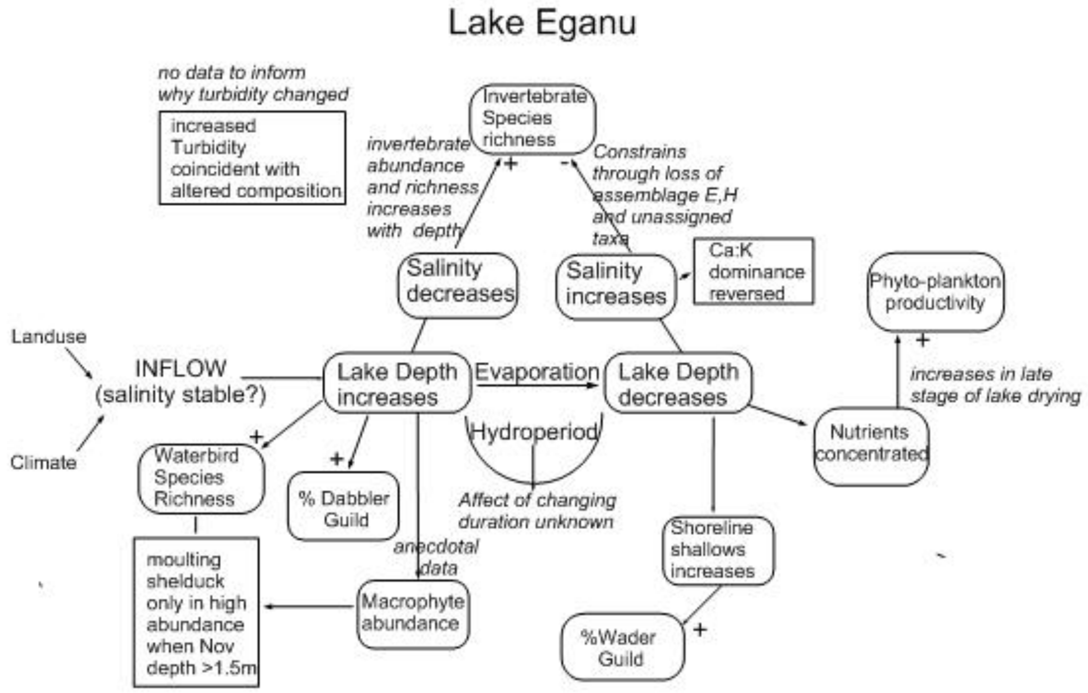


*Figure 24. Conceptual model of water body biodiversity components and their interaction with conditions at Lake Eganu.*

**Vegetation of the riparian zone.**

Lyons *et al.* (2007) documented historical changes to the vegetation of Lake Eganu, including the total collapse of the emergent vegetation of the lake basin. The loss of emergent basin vegetation has been widespread in response to salinisation and increased hydroperiods at wetlands across much of the Wheatbelt. A common outcome is the contraction of basin and riparian tree species to a narrow zone. Few studies have documented whether, following the loss of basin vegetation, the remaining narrowed riparian zone continues to decline or remains stable. Wetlands that have experience prolonged flooding (leading to the loss of the lake floor vegetation) yet remain relatively fresh, appear to retain a narrowed zone of healthy riparian vegetation and experience episodic recruitment events following large fill events (see Froend & van der Moezel, 1994, Lyons unpublished data). At secondarily salinised wetlands, the outcomes for fringing vegetation are likely to depend on a number of factors including soil type, groundwater salinity and depth, riparian topography and the frequency and duration of larger wetland fill events (along with water body salinity) that inundate the riparian zone.

Evidence from the current study demonstrates that the riparian zone vegetation continues to decline at Lake Eganu, following the loss of the lake bed vegetation. However stands of low condition rating are persisting. Importantly no recruitment has occurred over almost a decade of monitoring despite a number of years with significant rainfall. Vegetation persistence at transects, and most likely the entire lake shore, is spatially variable and probably responds to depth and salinity of groundwater, soil salinity and soil type along with inundation in areas close to the wetland. Isolating the critical drivers of persistence or decline and the lack of recruitment would require further detailed hydrological and ecological studies. The decline in condition of *M. strobophylla* within the upstream subsidiary wetland coupled with the observed collapse of stands within nearby basin wetlands suggests that the process of salinisation is continuing within the drainage line upstream of Lake Eganu.

Trends in the condition of the riparian vegetation at Lake Eganu may contrast with the biodiversity values of the water body under scenarios of repeated large fill events. While major fill events promote the persistence of the invertebrate communities and waterbird usage of the lake, as discussed above, they are likely to further degrade the riparian vegetation. Filling corresponds with elevated saline groundwater and potential surface inundation that will transport salt loads into the riparian zone. In contrast, moderate rainfall events may ameliorate soil salinities within the riparian zone yet be insufficient to fill the lake. Although the lake's salt load appears to be stable, any increase, particularly if coupled with fill events that inundate the riparian zone or lead to very shallow groundwater, is likely to lead to accelerated riparian vegetation decline and further degrade the water body for aquatic invertebrates and waterbirds.

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## APPENDICES

***Appendix A.*** *Water chemistry parameters for Site A at Lake Eganu 1998-2006 monitoring years.*

| **Date** | **20/08/98** | **12/11/98** | **11/05/99** | **22/08/00** | **9/11/00** | **13/02/01** | **4/09/02** | **16/10/02** | **26/03/03** | **27/08/04** | **1/11/04** | **5/04/05** | **11/09/06** | **17/10/06** | **1/03/07** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Monitoring Year | 1998 | 1998 | 1998 | 2000 | 2000 | 2000 | 2002 | 2002 | 2002 | 2004 | 2004 | 2004 | 2006 | 2006 | 2006 |
| Depth (m at gauge) | 2.56 | 2.15 | 1.8 | 1.74 | 1.58 | -9999 | 0.7 | 0.61 | Dry | 2.44 | 2.18 | 0.98 | 0.79 | 0.69 | Dry |
| EC (μS/cm) | 26800 | 43300 | 55700 | 107800 | 128600 | 226000 | 236000 | 196000 | | 25100 | 49300 | 188700 | 180600 | 174800 | |
| pH | 8.12 | 9.01 | 8.25 | 8.04 | 7.81 | 7.3 | 7.56 | 7.13 | | 8.16 | 8.53 | 7.44 | 7.72 | 7.02 | |
| TFN(μg/L) | 1200 | 1900 | 2700 | 3400 | 4400 | 11000 | 5700 | 19000 | | 3400 | 1400 | 8200 | 11000 | 22000 | |
| TFP(μg/L) | 10 | 10 | 20 | 5 | 5 | 10 | 5 | 10 | | 10 | 10 | 50 | 90 | 170 | |
| Cphyll-a (μg/L) | 1 | 10 | 0.5 | 0.5 | 0.5 | 15 | 11 | 3 | | 2 | 0.5 | 0.5 | 0.5 | 9 | |
| Cphyll-b (μg/L) | 0.5 | | 0.5 | 0.5 | 0.5 | 5 | 4 | 0.5 | | 0.5 | 0.5 | 0.5 | 0.5 | 0.5 | |
| Cphyll-c (μg/L) | 0.5 | | 0.5 | 0.5 | 1 | 6 | 5 | 0.5 | | 0.5 | 0.5 | 0.5 | 0.5 | 5 | |
| Pphytin-a (μg/L) | 0.5 | | 7 | 0.5 | 0.5 | 0.5 | 1 | 0.5 | | 0.5 | 0.5 | 0.5 | 3 | 0.5 | |
| Temp (°C) | 16.9 | 28.3 | 18.8 | 14.1 | 29 | 34.1 | 22.1 | 32.3 | | 17.9 | 27.2 | 32.1 | 19.7 | 25.3 | |
| Diss.Oxy.(%) | 105 | 123 | | 54 | 76.4 | 42.5 | 96.2 | | | 99.8 | | 57.1 | 61.9 | -9999 | |
| Turbidity (NTU) | | 990 | | | 2.4 | | | 17 | | | 0.2 | | | 25 | |
| Colour (TCU) | | 13 | | | 33 | | | 29 | | | 11 | | | 10 | |
| TDS (g/L) | | 30 | | | 110 | | | 320 | | | 32 | | | 300 | |
| Alkalinity (mg/L) | | 180 | | | 210 | | | 655 | | | 141 | | | 675 | |
| Hardness (mg/L) | | 4600 | | | 15000 | | | 98000 | | | 4900 | | | 120000 | |
| Si (mg/L) | | 8 | | | 13 | | | 6.1 | | | 5.6 | | | 2.8 | |
| Na (mg/L) | | 8500 | | | 34200 | | | 90800 | | | 11000 | | | 93500 | |
| Ca (mg/L) | | 350 | | | 1550 | | | 338 | | | 477 | | | 292 | |

***Appendix A.*** *Water chemistry parameters for Site A at Lake Eganu 1998-2006 monitoring years.*

| **Date** | **20/08/98** | **12/11/98** | **11/05/99** | **22/08/00** | **9/11/00** | **13/02/01** | **4/09/02** | **16/10/02** | **26/03/03** | **27/08/04** | **1/11/04** | **5/04/05** | **11/09/06** | **17/10/06** | **1/03/07** |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Monitoring Year | 1998 | 1998 | 1998 | 2000 | 2000 | 2000 | 2002 | 2002 | 2002 | 2004 | 2004 | 2004 | 2006 | 2006 | 2006 |
| Mg (mg/L) | | 880 | | | 2750 | | | 23700 | | | 911 | | | 28100 | |
| K (mg/L) | | 200 | | | 594 | | | 4330 | | | 200 | | | 4540 | |
| Mn (mg/L) | | 0.05 | | | 0.025 | | | 0.87 | | | 0.009 | | | | |
| Cl (mg/L) | | 17000 | | | 56000 | | | 189000 | | | 19000 | | | 185000 | |
| $HCO_3$ (mg/L) | | 160 | | | 256 | | | 799 | | | 121 | | | 824 | |
| $CO_3$ (mg/L) | | 30 | | | 1 | | | 1 | | | 25 | | | 1 | |
| $NO_3$ (mg/L) | | 0.01 | 0.01 | 0.02 | 0.02 | 0.01 | 0.05 | 0.13 | -9999 | 0.01 | 0.005 | 0.005 | | 0.06 | |
| $SO_4$ (mg/L) | | 1800 | | | 6850 | | | 29000 | | | 186 | | | 30100 | |