State Salinity Strategy Wetland Biodiversity Monitoring Report:

Lake Wheatfield 1997 to 2009

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SUMMARY

Lake Wheatfield is part of a series of naturally saline wetlands forming the Warden Wetlands system. The Warden wetlands are managed under the RAMSAR convention on the basis of importance to migratory waders and south-west populations of Hooded Plover and Chestnut Teal. Lake Wheatfield, like other wetlands in the system, has been affected by altered hydrology (increased lake depth and extended duration of inundation) as a result of clearing for agriculture. An engineering solution to reduce depths within the central suite of wetlands (including Lake Wheatfield) has been installed and has been operating since early 2009.

Lake Wheatfield was included in the State Salinity Strategy Biodiversity Monitoring Program because; it has a high conservation value, is actively managed as part of the Warden Natural Biodiversity Recovery Catchment and is an example of a naturally saline wetland. Data collection commenced in 1997 and is ongoing. In this report, an interpretation of data collected from 1997 – 2009 is presented for water chemistry, riparian vegetation, waterbirds and aquatic invertebrates.

Groundwater levels (AHD) in observation bores associated with vegetation monitoring transects increased over the monitoring period. Average lake depth was 1.6 m and rarely fell below 1m. Consequently, two dominant species amongst the riparian vegetation, *Melaleuca cuticularis* and *Eucalyptus incrassata,* have shown a decline in basal area and canopy condition over the monitoring period. The declining condition of these species has occurred at low elevations and is likely to be a direct result of salinity and prolonged inundation. No such decline was observed at higher elevations.

Forty two waterbird species were recorded, of which eight were observed breeding. While species richness was high, Lake Wheatfield supported only a small and variable portion of the total abundance of birds in the wider Warden system. Species richness was greater during our surveys than during surveys conducted in the 1980s, but it is suggested that this is largely related to survey methods used since there are few changes in cumulative species lists between these periods. Lake Wheatfield is not an important site for Hooded Plover, but Chestnut Teal were present in all surveys. Chestnut Teal occurred at Lake Wheatfield with variable abundance, but usually as only a small portion of the local abundance in the Esperance region.

One hundred and thirty three invertebrate species were collected, including a small number of rarely encountered species. Invertebrate richness and community structure was variable from year to year, but there was little directional change that would indicate a change in condition; rather, most species are likely to have occurred in the lake in most years, but not always within the sampling window. Samples in spring 2009 had lowest invertebrate richness, coinciding with lower depth following the engineering works. Monitoring in coming years will determine whether lowered depth is a sustained influence on invertebrate communities.

It is suggested that the need for seasonally lower depth in Lake Wheatfield is confirmed by the loss of condition in riparian vegetation. Lower water depths will be beneficial to most waterbirds in the long term with the possible exception of reduced nesting areas for ibis and spoonbills which have benefited from stands of flooded

trees. Lower water depths will not conflict with the management of invertebrate communities provided a similar seasonal range of depth and salinity is maintained. The gravity fed pipeline to reduce water levels in Lake Wheatfield and downstream wetlands, was installed at the end of the monitoring period reported here. Our data thus form an important baseline for monitoring the future effects of the intervention on the wetland's biodiversity.

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, shallow monitoring bores, and salinity of riparian soils. A detailed description of the monitoring protocol is given by Cale *et al.* (2004) and Gurner *et al.* (1999) 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 that will analyse and interpret the data collected for individual wetlands within the State Salinity Strategy Wetland 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	√	√		√	√	√
Towerrinning	Improving		√		√	√	
Walyormouring	Secondary saline		√	√	√		
Wheatfield	Primary saline	√	√	√		√	
Yaalup	Declining		√	√	√		√

INTRODUCTION

Lake Wheatfield (33° 48' 121° 55') is part of a complex of wetlands forming the Lake Warden Wetlands on the northern edge of the town of Esperance on the south coast of Western Australia. The complex of wetlands covers an area of approximately 2000 ha and includes 8 major lakes and 90 satellite overflow wetlands (DEC 2009), divided into western, central and eastern suites. The Lake Warden Wetlands were listed under the RAMSAR convention in 1990 because they were; a) unique in character for the region, b) an important drought refuge for south-west waterbirds and a stopover site for migratory waders and c) support in excess of 1 % of the total population of Chestnut Teal and Hooded Plover occurring in the south west region. Lake Wheatfield has consistently been the most important wetland, in the larger complex, for Chestnut Teal (DEC 2009, Pinder *et al.* 2010).

Surface and groundwater hydrology of the Lake Warden Wetlands have been described previously (DEC unpublished Draft Recovery Catchment plan 2005) and are summarized below. Lake Wheatfield is part of the central suite of Warden wetlands. Surface drainage to this suite is from Coramup Creek which has a catchment area of 39,480 ha and between 1998 and 2003 had an annual mean discharge of 5.1 x 10^6 m^3. Coramup Creek enters Lake Wheatfield directly and water flow continues through the suite from east to west entering Woody Lake through a well defined channel and thence into Lake Windabout and finally into Lake Warden; the latter a part of the western suite of wetlands. Outflow from Lake Wheatfield, east across Fisheries Road to Bandy Creek, may occur when the lake fills to AHD 4.8 m (1.77 m at the depth gauge). The direction of flow from Lake Warden may reverse back toward Lake Wheatfield when inflow from Coramup Creek is low, Lake Warden water levels are high (4.8 m AHD) and overflow is occurring at Fisheries road into Bandy Creek.

Lake Wheatfield is naturally saline with salinity ranging from 4 to 15 ppt since recording began in 1997 (see DEC 2009 and Lane 2008). Salinity data indicate a gradient through the central suite of wetlands with mean salinities (± standard deviation) of 6.86 (± 2.2) ppt, 7.37 (±2.69) ppt and 10.82 (± 3.86) at Wheatfield, Woody and Windabout lakes respectively (DEC 2009). The Eastern suite tends to be more saline although Ewans Lake may at times have salinities comparable to the central suite (DEC 2009). The Western suite is generally hypersaline with Lake Warden expressing salinities as low as 15ppt only at great depth (Lane 2004, DEC 2009).

There is a rapid decline in rainfall northwards away from the coast with average annual rainfall at the lakes in the order of 600 mm compared to 350 mm in the upper catchment. The majority of rain falls between June and August. Summer rainfall is generally low, however, there has recently been several occasions of extreme rainfall events during summer, i.e. 1999, 2000, 2007 and 2009, with falls in excess of 100 mm occurring in a short period of time and flooding the Wetland chain and satellite wetlands.

The catchment of Coramup Creek has been extensively cleared for agriculture and this has resulted in increased discharge into Lake Wheatfield. It has been estimated that up to 60% of surface flow in Coramup Creek is derived from ground water seepage from the catchment (DEC unpublished Draft Recovery Catchment plan 2005). Coramup Creek has also been identified as the major source of nutrients (nitrogen and phosphorus) into Lake Wheatfield.

Lake depths in the Warden Wetlands have trebled since 1985 (DEC Rec catch doc) and this is the main threatening process to wetland biodiversity. When discharge to Bandy Creek ceases, evaporation and groundwater recharge are the only processes acting to reduce water levels. This has resulted in an increased period of inundation and reduced areas of shallow water and exposed shoreline which have had impacts on riparian vegetation and waterbird diversity (e.g. Robertson and Massenbauer 2005).

Early on-ground management in the catchment focused on targeted planting of perennial species within agricultural zones of the catchment, in order to reduce inflow and salt load to the Warden Wetland system. In anticipation that this would be insufficient to address the altered hydrology of the wetland system a management plan involving 5 components was developed simultaneously (DEC 2005, Maunsell 2006). The first component (component 1 option 1A; Maunsell 2006) of this plan was completed in April 2009 and included a pipeline to take water from Lake Wheatfield at depths greater than 1.7m (3.8m AHD) and discharge into Bandy Creek, thus reducing the volume of inflow to downstream wetlands. Initial evaluation suggests that the pipeline effectively reduced water levels in not only Lake Wheatfield, but also Lakes Woody and Windabout, resulting in increased areas of exposed shore (pers. comm. John Lizamore DEC Esperance).

Lake Wheatfield was originally included in the monitoring program because it met five of the seven design criteria (Cale *et al.* 2004). Lake Wheatfield; 1) was listed in the original salinity Action Plan document (Govt Aust 1996) and is now managed by the Department of Environment and Conservation as a Natural Diversity Recovery Catchment (DEC 2005) 2) fits within the program design as a naturally saline wetland with declining quality, 3) had high conservation value for waterbirds (RAMSAR listing) and vegetation, 4) extended the representation of the variety of wheatbelt wetlands, 5) was subject to active catchment management.

The ecological character description for The Lake Warden Wetlands (DEC 2009) thoroughly reviews the various components of the ecosystem and indicates the standing of Lake Wheatfield as a diverse, moderately productive wetland within the greater wetlands system.

Data for aquatic invertebrates, from the State Salinity Strategy Wetland Biodiversity Monitoring Program, have previously been reported up to 2000 (Cale *et al.* 2004) and indicated a relatively high species richness (52-73 species) dominated by insects (40-45% of species). A significant marine influence on the fauna was suggested by the presence of species such as *Melita kauerti* (Amphipoda), and *Exosphaeroma* sp. (Isopoda).

Recent sampling has determined an average total biomass of the invertebrate community of 0.14 g in 2006 (Cook *et al.* 2006), 0.09 g in 2007 (Cook and Farrell 2008) and 0.40 g in 2009 (Pinder *et al.* 2010) for a qualitative sampling path 50m long. Low biomass recorded in 2007 coincided with a shift of community structure from dominated by macroinvertebrates in 2006, to equal biomass of micro and macro invertebrates in 2007. Sampling by Cook et al. (2006), Cook and Farrell (2008) and Pinder *et al*. (2010) suggested that Lake Wheatfield had high invertebrate diversity at least as high as several nearby wetlands.

Waterbirds at Lake Wheatfield have previously been surveyed as part of surveys of the Warden wetlands more generally (Jaensch *et al.* 1988, Clarke and Lane 2003, Halse 2007, Halse 2008, Pinder *et al.* 2010). These surveys reported between 17 and 26 species and indicate that only a small portion of the abundance distributed across the Warden wetlands occurs on Lake Wheatfield. Aerial survey of the Warden wetlands and the Gore-Quallilup system (Halse 2007, Halse 2008, Pinder *et al.* 2010) indicated that while Lake Wheatfield supported only a small portion of the abundance, species richness was greater than in most other wetlands of either system.

Analysis of the effect of increased water depth on waterbird diversity at Lake Warden (Robertson and Massenbauer 2005) revealed a negative relationship and identified an annual depth range of 0.3 – 1.3 m to optimise diversity for that lake. The depth diversity relationship was weaker at Lake Wheatfield because less habitat was lost with increased depth and analysed data included a high proportion of diving species with little reliance on shallow habitats.

The riparian vegetation of Lake Wheatfield was sampled as part of a major floristic survey of the Western Australian agricultural zone and was typical of the coastal and near coastal wetlands of the south coast (Lyons *et al.* 2004, Halse *et al.* 2004) being dominated by *Melaleuca cuticularis*. Vegetation monitoring commenced in 1997, at which time the trees of the littoral zone were observed to be showing some signs of stress (Gurner *et. al.* 1998).

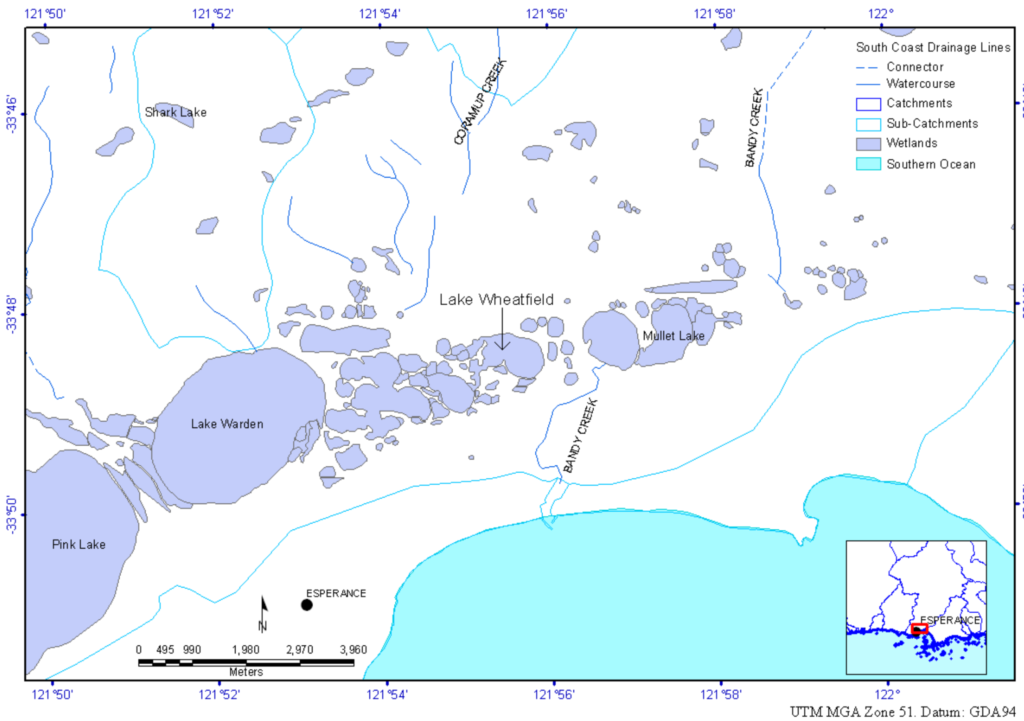

Figure 1. Catchment setting of Lake Wheatfield.

METHODS

The data reported here were 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 was in the vicinity of the bird hide on the lake's south western shore, while site B, selected to sample a different quadrant of the wetland, lies on a shallow vegetated spit marking the edge of the channel between the lakes southern and central basins. Sub-sites have no relevance to waterbird data which were collected over the whole wetland.

Table 2. Location of vegetation transects and groundwater monitoring bores. Transect coordinates are for the transect corner pegs. mCALM values are the vertical height above the lowest point of the lake (i.e. equivalent to lake depth). Horizontal values (eastings and northings) are approximate MGA 94 (+/- 10m).

Feature	NORTH (MGA94) Zone 51	EAST (MGA94) Zone 51	mCALM	AHD HT
TRANSECT 1				
1A	6258841.6	401020.6	0.90	3.93
1B	6258854.5	401035.6	2.09	5.12
1C	6258882.8	401038.6	2.11	5.14
1D	6258867.3	401050.9	1.07	4.1
1E	6258857.3	401007.9	0.90	3.93
1F	6258870.1	401023.3	2.50	5.53
TRANSECT 2				
2A	6258673.9	401113.9	0.78	3.81
2B	6258678.7	401133.2	2.91	5.94
2C	6258683.7	401152.5	4.23	7.26
2D	6258654.6	401118.5	0.80	3.83
2E	6258659.3	401137.7	3.21	6.24
2F	6258664.3	401157.1	3.89	6.92
TRANSECT 3				
3A	6258280.4	400554.9	1.27	4.3
3B	6258297.0	400566.1	1.14	4.17
3C	6258313.5	400577.2	0.66	3.69
3D	6258269.3	400571.2	1.61	4.64
3E	6258285.8	400582.5	1.11	4.14
3F	6258302.3	400593.8	0.76	3.79
TRANSECT 4				
4A	6258854.2	400271.2	2.43	5.46
4B	6258834.8	400266.7	2.30	5.33
4C	6258839.3	400247.3	2.22	5.25
4D	6258858.7	400251.9	2.48	5.51
4E	not marked		0.99	4.02
4F	not marked		0.96	3.99
Monitoring Bores				
LW B 001	6258898.0	401074.6	1.97	5.0
LW B 002	6258902.1	401104.1	3.08	6.11
LW B 003	6258697.1	401186.9	4.24	7.28
LW B 004	6258682.9	401154.4	3.87	6.90
LW B 005	6258221.7	400997.0	2.35	5.38

Vegetation

Four transects (all 40 m long) were positioned around the lake to sample representative stands of the different vegetation communities, elevations and topographic positions. Transects were orientated at right angles to the prevailing slope and correspondingly to the wetland edge.

Transect 1 (Fig. 2) sampled *Melaleuca cuticularis, Melaleuca. brevifolia* and *Spyridium globulosum* near the inflow creek which the transect crosses, then back into terrestrial vegetation with an overstorey of *Acacia saligna,* and then into the wetlands edge again with *M. cuticularis* and *Sarcocornia quinqueflora* with mixed sedges at lower elevations. It is located on the north-eastern side of the lake approximately 30 m west of the car park and extends from the terrestrial vegetation to the lake edge.

Transect 2 (Fig. 2) sampled *Banksia speciosa* woodland over mixed myrtaceous species and *Darwinia diosmoides* on a dune rise, extending downslope to *Melaleuca cuticularis* woodland over scattered sedges including *Ficinia nodosa* and *Baumea juncea* in the littoral zone of the wetland. It is located on the eastern side of the lake, approximately 50 m south-east of the car park.

Transect 3 (Fig. 2) sampled mature *Melaleuca cuticularis* woodland and is located on the southern side of the lake. All but the upper edge of the transect is located in the sub-littoral zone of the wetland and has been inundated for the duration of the study.

Transect 4 (Fig. 2) comprises open woodland of *Eucalyptus incrassata* and scattered *E. occidentalis* over *Leucopogon revolutus / Labichea lanceolata* over *Baumea juncea.* This transect is located off the north-west side of the wetland and samples and samples an island created by two branches of the outflow channel channel.

Timing of sampling

Invertebrates and waterbirds

The aquatic fauna at Lake Wheatfield has been sampled every second year from 1997 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 water levels are at their lowest. These sampling events span two calendar years, but, 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. Where year is used in the text below it refers to these monitoring years. To denote the three sampling events capitalised season names are used, i.e. Late-winter 1998 is the sample from Aug-Sep 1998 while Autumn 1998 would be the sample collected in Mar-Apr 1999. In some years, a wetland will fail to fit neatly with the conceptual sampling regime because either summer rainfall results in peak wetland volumes after the spring sampling or winter inflows fail to occur. Consequently, samples from any monitoring year must be recognised as a snap shot of conditions at a (sometimes unknown) point along the trajectory of a hydrological cycle.

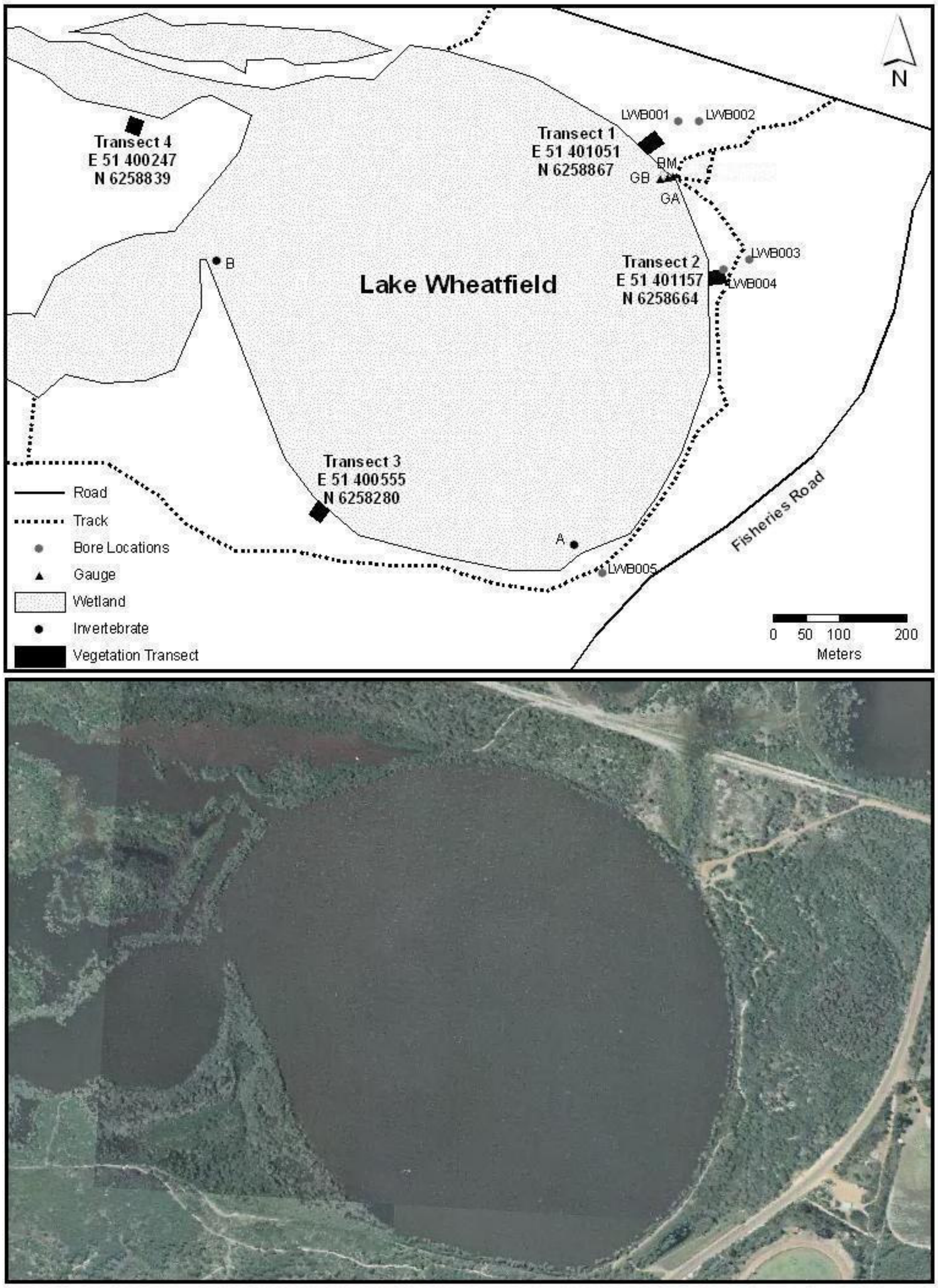

Figure 2. Location of sampling sites at Lake Wheatfield. (GA - depth gauge A, GB –depth gauge B, A – invertebrate sub-site A, B – invertebrate sub-site B, BM – survey bench mark). Vegetation transect coordinates are for the upper left corner peg (as seen when facing the wetland). Shallow groundwater observation bores are shown (LWB001-LWB005).

Vegetation, shallow groundwater and soil conductivity.

Monitoring of the vegetation commenced in 1997, with sampling repeated in 2000, 2003, 2006 and 2009. Field measurements occurred in November or December in each of the five sampling years, with the exception of transects 1, 2 and 3 in the 2003 sampling round which were sampled in May 2004. Soil electrical conductivity measurements were 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.

Five shallow groundwater monitoring bores located closest to the vegetation transects were selected from a network of 13 bores established in close proximity to Lake Wheatfield. Additional bores were located elsewhere in the catchment. Monthly depth to groundwater measurements in the vicinity of Lake Wheatfield commenced in September 2000. Groundwater measurements were taken frequently to maximize the probability of capturing seasonal variation and the effect of pumping.

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 the permanent gauge installed in November 1999 as part of the South-West Wetland Monitoring Program (Lane *et al.* 2008), prior to this date depth was estimated, 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 μm 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 three sampling occasions within a monitoring year. All individuals of all obligate wetland species (includes species such as Swamp Harrier and White-bellied Sea Eagle) were identified and counted using binoculars or a spotting scope. As much of the wetland was traversed as was practical; by boat where water levels were sufficient or on foot at lower depths. Except when water levels were high the surveyed area was defined by the contiguous wetted area, however, at high water levels the surveyed area ended at the boundary with Woody Lake (i.e. it included the channel between Lake Wheatfield and Woody Lake).

To facilitate interpretation of waterbird community structure species were placed into one of eight feeding guilds. These guilds and species assignments follow Halse (1983) and use the means of foraging, e.g. dabbling, diving or wading (where large and small reflect deep and shallow feeding habitats) and the principal food type i.e. animal, mixed (as in dabblers) or vegetation. Shore feeding species generally feed peripheral to the wetland, e.g. ibis and Banded Lapwing; they were not separated by food type. Aerial feeding species include the raptors and terns. An additional guild was erected to reflect the specialised foraging in reed habitats of species such as the Little Grassbird, Clamorous Reed-warbler and Baillon's Crake.

Invertebrate sampling

Invertebrate samples were collected during Spring sampling events each monitoring year. The invertebrate sampling protocol focused 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 comprised two sub-samples collected using D-framed pond nets with intermediate (250 µm) and fine (50 µm) mesh sizes. The two sub-samples were collected in order to increase the efficiency with which both macro- and microinvertebrates were 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 200 m radius boundary) to a depth of 1.0 m. 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 2010) 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. Several dipteran families (Dolichopodae, Tabanidae, Tipulidae and Muscidae) were identified to family level only and Turbellaria, Nematoda, Mestostigmata and Oribatida were 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. Annual species and family richness, and assemblage composition, were calculated using species lists corrected for both sub-sites on a sampling date within the wetland.

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.

Vegetation condition and population structure

The detailed vegetation methods outlined in Lyons *et al.* (2007) are largely restated here. Transects 1, 2, 3 and 4 consisted of two contiguous 20 x 20 m 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 understory measurements as well as throughout the transect.

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 were calculated for the period 1997 to 2009 at the subplot scale (4 x 20 m) for plants >20mm DBH. Size class distributions were plotted for dominant species at each transect for the five 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. Elevation data were transformed to height above the lowest point of the wetland and transformed to mAHD. These data enable the examination of the inundation history of transects during wetland fill events by linking them to wetland depth gauge data.

Vegetation transects -shallow groundwater monitoring

Shallow groundwater monitoring bores were established near, but not adjacent to, vegetation transects as per other SAP monitoring wetlands. Observation bores depths ranged from 5.0 m to 7.275 m AHD.

Vegetation transects - soil electrical conductivity

Measurements of soil apparent electrical conductivity (ECa 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 round. 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, or one metre outside if there were fallen logs or other obstructions, at peg locations.

Invertebrate and waterbird data analysis

Species Accumulation

The accumulation of invertebrate species with repeated sampling was analysed using the EstimateS package (Colwell 2006). To estimate the potential total richness of the wetland species pool within EstimateS, the incidence based estimate of species richness, Chao2 (see also Chao 2005), with the 'classic' option was calculated using the frequency of occurrence of a species at the 2 sub-sites (i.e. collected at 1 or 2 sub-sites).

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. In general MST connects pairs of samples with the greatest similarity. Flexible UPGMA classification (PATN) was used to determine group membership of samples after ordination with the number of groups derived approximating 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 Wheatfield 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). In the absence of marker wetlands an ordination of samples from Lake Wheatfield would fill the ordination space even though between sample differences were small. Since each marker wetland is considered typical of a different wetland type the differences between them are relatively large. This establishes an ecological scale, with small annual changes in species composition occupying only small portions of the ordination space. Marker wetlands have the added benefit of indicating the types of wetlands to which Lake Wheatfield is more similar. While it is recognised that further investigation is required to refine the choice of marker wetlands, those used by Cale *et al.* (2004) are reused here. Lake Toolibin (data collected in 1983 by Jaensch *et al.* 1988) is an example of a 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 Pinjareega (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.

For invertebrate analyses, the same rationale was applied when using marker wetlands. Four marker wetlands with disparate invertebrate assemblages were selected by Cale *et al.* (2004) and are re-used here. Lake Campion has a species poor fauna (sampled in Oct 1998) of insects and Crustacea typical of a large saline wetland. Noobijup Swamp is a perennial, freshwater, reed swamp and supported (when sampled in Oct 1998) a highly diverse species assemblage, including both macro and micro invertebrates from a broad suite of higher taxa. Yaalup Lagoon is an ephemeral, brackish wetland (sampled in Oct 1999) with a diverse invertebrate assemblage dominated by insects. Lake Parkeyerring is a secondarily salinised wetland with a species assemblage (collected in Oct 1999) of low species richness and crustacean dominance.

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. Where appropriate, Kruskal-Wallis ratios (Belbin and Collins 2008) are reported to identify PCC vectors with the greatest discrimination between groups of samples following ordination.

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 distributions (as a surrogate for age structure) of stands and an assessment of recruitment, and growth or decline of mature stands.

Percentage changes in basal area from 1997 to 2009 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 and transect inundation history. The decline or accumulation of basal area (decline – due to death or stress, accumulation due to growth and/or recruitment) was also examined by linear regression against apparent soil conductivity ECa.

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 times 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 *M. cuticularis*, *M. brevifolia* and *Eucalyptus incrassata* at the transect scale. Wilcoxon matched pairs test (Statsoft, 2001) for dependent samples was also performed, comparing 1997 to 2009 data.

RESULTS

Water chemistry

Depth

The Lake Wheatfield depth gauge was installed in November 1999 (Lane *et al.* 2008). Lake depths before this time were estimated on-site and while indicative of conditions may be over estimates, at least at higher depths.

Lake Wheatfield was permanent over the study period with actual minimum annual depths likely to be within 10 cm of the minimum depths recorded in this study (based on more extensive data collected by DEC Esperance). The spring maximum and autumn minimum means were averaged for the period 2000 – 2009 which gave a mean lake depth of 1.6 m (standard deviation = 0.2 m). Water depth rarely fell below 1m; the lowest depth recorded during water bird and invertebrate sampling occurred in Autumn[1] 1997 when levels dropped to 0.7m, followed by Autumn 2009 when levels were 0.82m after the installation of the drainage pipeline (Fig. 3). In periods between fauna sampling water levels fell below 1m in Feb-Apr 2003 and Feb 2005 (DEC Esperance unpublished data). Lake water level fluctuations were associated with month to month variation in rainfall in Esperance, presented as the cumulative deviation from the mean (CDFM) (Fig. 3b).

Maximum recorded depth usually occurred in Spring, except in 2009 when depth peaked prior to Spring and was then reduced by the operation of the drainage pipeline (Fig. 5). In all years the difference between Late-winter and Spring depths was less than 0.36 m (mean = 0.13, sd= 0.12). The lake reached overflow (>1.8m) depths in Oct 2003, Jun 2005, Jan and Oct 2007 and Aug 2009 (DEC Esperance unpublished data). Depths > 1.45m at the depth gauge inundated significant portions of vegetation transect 3 and lower sections of transects 1, 2 and 4.

[1] Capitalised seasons refer to sampling events during a monitoring year; Autumn 1997 was sampled in March 1998, see Methods for more detail.

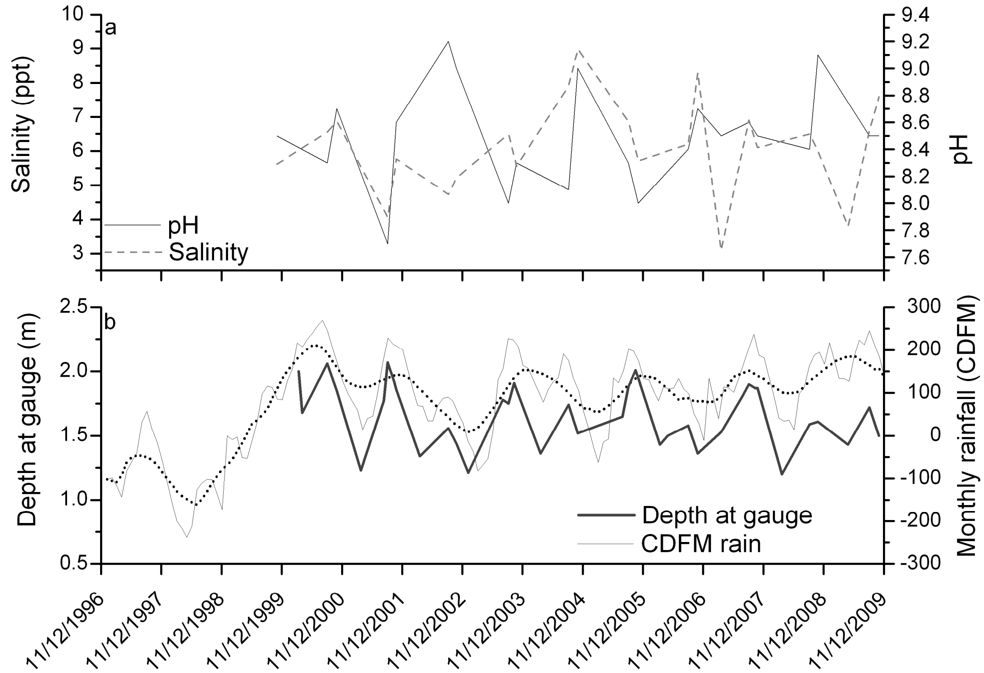

Figure 3. Lake Wheatfield water salinity and pH (a), and standing water depth and monthly rainfall (as cumulative deviation from the mean - CDFM) for Esperance (b) over the monitoring period. The dotted line is a 10-point running mean of CDFM.

Salinity

Salinity (as indicated by electrical conductivity) was negatively correlated with water depth ($r=-0.847$, $df = 17$, $p<0.01$, see Fig. 4). The lowest recorded salinity was 505 mS/m in Autumn 1999 and coincided with high but declining water levels. The maximum salinity was 2440 mS/m recorded in Autumn 2009. During a monitoring year salinity tended to peak in autumn after summer evaporation had reduced water levels (Fig. 5). Spring salinity was within the range 901 - 1261 mS/m. At depths > 1.45 m, when vegetation transects were significantly inundated, electrical conductivity ranged from 649 mS/m to 756 mS/m.

A cation dominance hierarchy of Na>Mg>Ca>K was observed throughout the study and chloride was the dominant anion. Carbonate was at or below the lower detection limit (1 mg/l) in all samples except Spring 2007 when it was at higher concentrations than bicarbonate.

pH

The mean pH recorded during invertebrate and waterbird sampling was 8.3 but pH varied both seasonally and across years with a minimum of 6.69 (Late-Winter 2003) and a maximum of 9.35 (Spring 2009). The largest annual variation was 1.69 pH units in 2003 (range 6.69-8.33). Generally, pH tended to increase between late-winter and spring, but showed no consistent trend between spring and autumn (Fig. 5). At depths > 1.45 m, when vegetation transects were significantly inundated, pH was in the range 7.7 - 9.2.

There was a positive correlation between pH and two parameters (Fig. 4); EC ($r= 0.491$, $df = 16$, $p < 0.05$) and temperature ($r = 0.523$, $df = 15$, $p < 0.05$) on invertebrate and waterbird sampling occasions, however, more extensive data (Fig. 3a) show that any relationship with salinity was not simple and frequently did not hold. A negative correlation was observed between pH and colour ($r = -0.825$, $df = 5$, $p < 0.05$). Colour ranged from 32 – 160 TCU (which is quite low) and was not correlated with any other measured parameters.

Nutrients

Total soluble nitrogen (TFN) concentration was positively correlated with total soluble phosphorus (TFP) concentration ($r = 0.68$, $df = 16$, $p < 0.05$), however there was no significant correlation between either nutrient and photosynthetic pigments, i.e TFN and total chlorophyll ($r = 0.24$, $df = 16$, $p > 0.05$); TFN and Phaeophytin ($r = 0.31$, $df = 16$, $p > 0.05$; Fig. 4); TFP and total chlorophyll ($r = -0.16$, $df = 16$, $p > 0.05$;TFP and Phaeophytin ($r = -0.02$, $df = 16$, $p > 0.05$).

Where TFN and TFP were measured at two sub sites, i.e. in spring, occasional discrepancies between sites were substantial and indicated that nutrient concentrations may have been patchily distributed across the wetland (Fig. 6). For example total soluble phosphorus concentration was generally in the range <10 to 50 µg/l, however, in Spring 2007 a very high concentration of 700 µg/l was recorded at sub-site A compared to 20 µg/l at sub-site B. In Late-winter 2009, 480 µg/l of TFP was recorded from a single sample. Similar discrepancies were recorded for TFN which had a mean value of 1924 µg/l and a range of 1200- 4600 µg/l (Fig. 6).

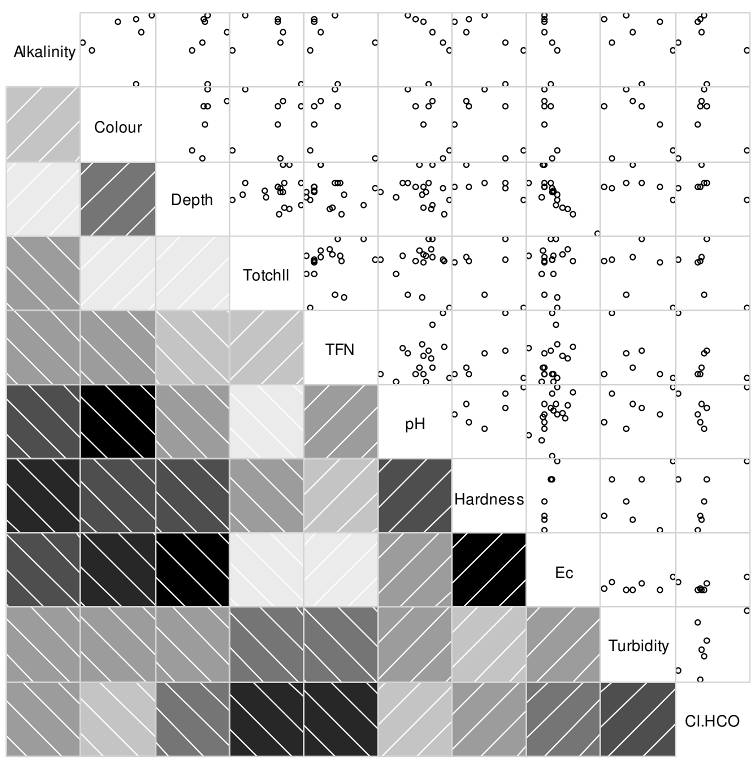

Figure 4. Correlation matrix for water chemistry parameters (diagonal) collected on all biological sampling dates. Darkest colours (lower panel) represent largest r values, and data values are plotted in upper panel. CL.HCO is the ratio of Chloride to Bicarbonate ions. Significant correlations are discussed in the text.

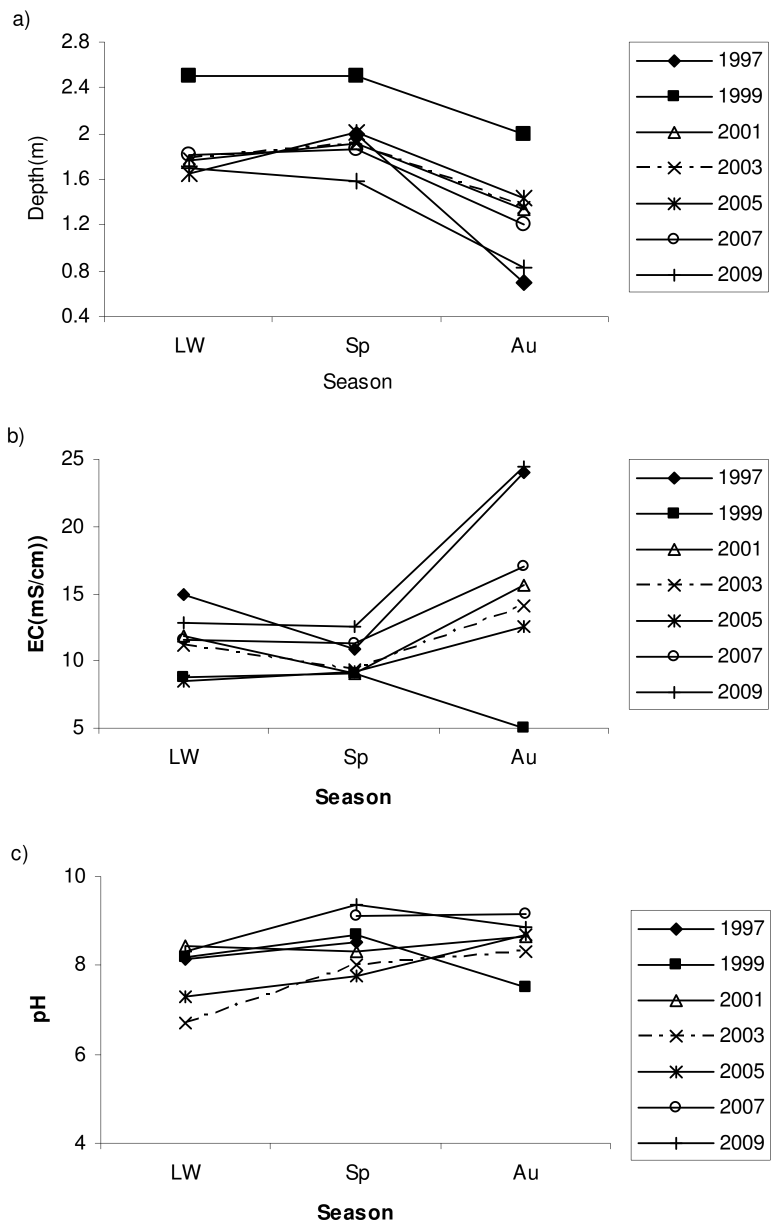

Figure 5. Seasonal values of depth, salinity and pH measured during waterbird and invertebrate sampling across the study period.

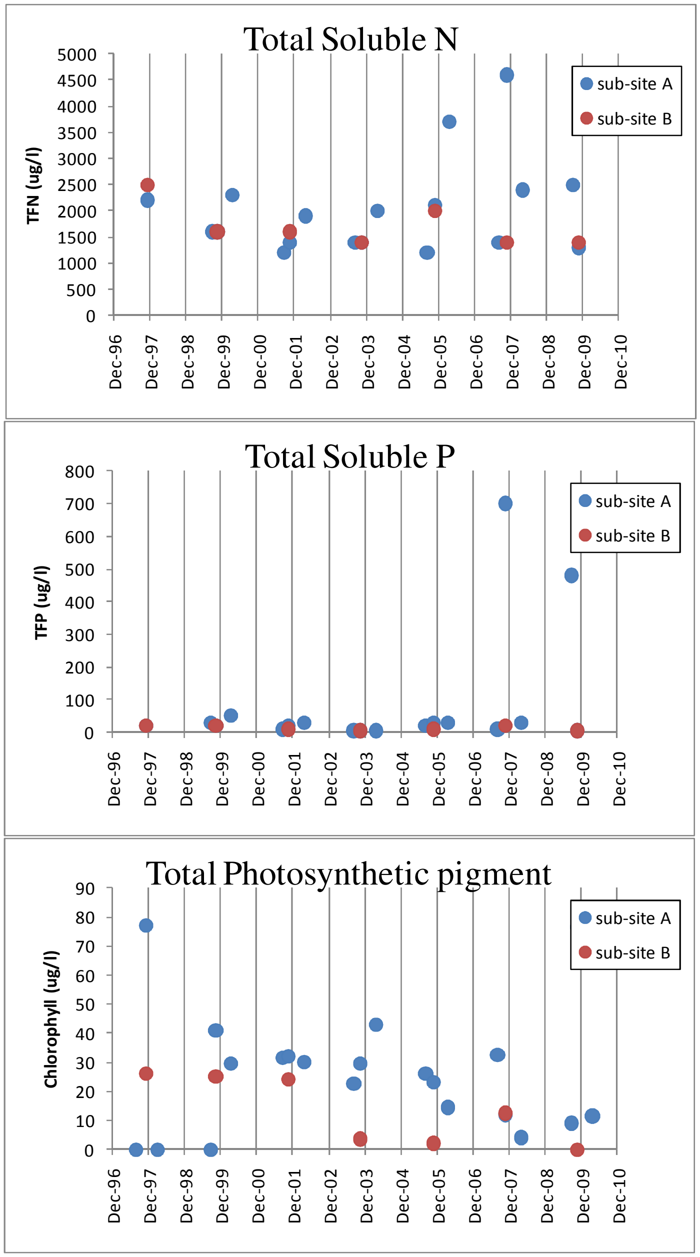

Figure 6. Total soluble nitrogen, total soluble phosphorus and total chlorophyll sampled during waterbird and invertebrate sampling across the study period.

Waterbirds

Description of waterbird assemblage

A total of 42 species of waterbird have been recorded during the monitoring period (Table 5). Four species, Chestnut Teal, Pacific Black Duck, Little Black Cormorant and Eurasian Coot, have been recorded on all sampling occasions and 11 species have been recorded on single occasions only. The Hooded Plover was not recorded for any surveys, although it is known to regularly occur on some wetlands in the larger Warden suite.

Eight species were recorded breeding during the monitoring period (Fig. 7). These included the communal nesting Yellow-billed Spoonbill, Darter, Straw-necked Ibis and Little Black Cormorant which are likely to have bred in large numbers in Spring 1997, 2005 and 2007, but for which nest abundance data was forgone in order to prevent excessive disturbance of nesting birds. There was evidence in most years of breeding amongst these species, but survey timing appeared to occur at nest building (Late-winter) and recently fledged (Spring) stages, thus reducing the number of recorded broods. Other breeding species included Grey Teal, Pink-eared Duck, Australasian Shoveler and Little Pied Cormorant.

The mean number of individuals (± standard deviation) counted per survey was 839 (± 938). Total abundance (log transformed) was significantly and negatively correlated with water depth ($r = -0.6$, $df = 17$, $p < 0.01$). This reflected a tendency for abundance to increase in Autumn[2] when birds tend to congregate on the permanent Lake Wheatfield following the summer drying of other temporary wetlands, rather than a general preference for shallower water. The greatest single count of birds was in Autumn 1997 when 4542 birds of 18 species were counted. There was no significant temporal trend in total abundance over time. (Regression adj $r^2 = 0.0636$, $F = 2.35$, $df = 19$, $p = 0.14$). Abundances of individual species did not show a temporal trend beyond a seasonal increase during autumn.

Abundances of individual species were generally higher in 1997 compared to subsequent years. For example, Chestnut Teal numbered 450 in Late-winter and 335 in Spring 1997 while on other sampling occasions abundance of this species was less than 100 (mean = 37.8 ± 22.9). Similarly, 1670 Pink-eared Duck, 1450 Grey Teal and 430 Australasian Shoveler were recorded in Autumn 1997, compared to means (for subsequent counts) of 114 ± 15, 148 ± 10 and 26 ± 16 respectively. Some species showed marked increases in abundance for individual surveys outside the 1997 monitoring year. For example 486 Straw-necked Ibis (mean = 44 ± 12 per survey) in Late-winter 2009 and 561 Pacific Black Duck (mean = 97 ± 6) in Autumn 2005. There was no significant temporal trend in total abundance (log transformed, all species) over the study period (Regression adj $r^2 = 0.0636$, $F = 2.35$, $df = 19$, $p = 0.14$).

Species richness (Fig. 7) was relatively constant compared to abundance with a mean of 20 ± 3 species per survey (range 14 - 26 species per survey). Species richness did not differ significantly between years (ANOVA, two factor without replication; $F(6, 12) = 1.72$, $p > 0.05$) or seasons ($F(2, 12) = 1.05$, $p > 0.05$). Species richness was not correlated with water depth ($r = -0.11$, $df = 21$, $p > 0.05$).

Total species richness was spread across 8 feeding guilds (Fig. 8). Dabblers, divers preferring animal prey and small waders were the most species rich guilds (9, 8 and 7 species respectively) and collectively accounted for 57% of species richness. The dabbler guild was generally the most abundant (19 to 84% of total abundance) although it was occasionally rivalled by the diver guilds. Dabbler abundance (log transformed) was negatively correlated with water depth ($r = -0.70$, $df = 18$, $p < 0.01$) but richness was un-correlated. Both richness and log transformed abundance in the small wader guild were negatively correlated with water depth ($r = -0.85$, $df = 18$, $p < 0.01$, and $r = -0.71$, $df = 18$, $p < 0.01$ respectively). Within the guild of divers feeding on vegetation, richness was positively correlated with depth ($r = 0.66$, $df = 18$, $p < 0.01$) and log transformed abundance was negatively correlated with depth ($r = -0.45$, $df = 18$, $p < 0.05$). So, as water depth decreased dabblers and waders became more abundant and, while waders became more diverse, vegetation eating divers became less diverse.

The waterbird community at Wheatfield bears little resemblance to those of the marker wetlands, but was most similar to those marker wetlands (Toolibin and Pinjarrega) with comparable richness (Fig. 9). Eight species recorded at Lake Wheatfield were not members of any marker-wetland community. These were, Great Crested Grebe, Australian Pelican,

[2] Capitalised seasons refer to sampling events during a monitoring year; Autumn was sampled in March - April, see Methods for more detail.

Common Greenshank, Black-fronted Dotterel, Straw-necked Ibis, Little Black Cormorant, Darter and Common Sandpiper. To some extent this reflects the near-coastal location of Lake Wheatfield.

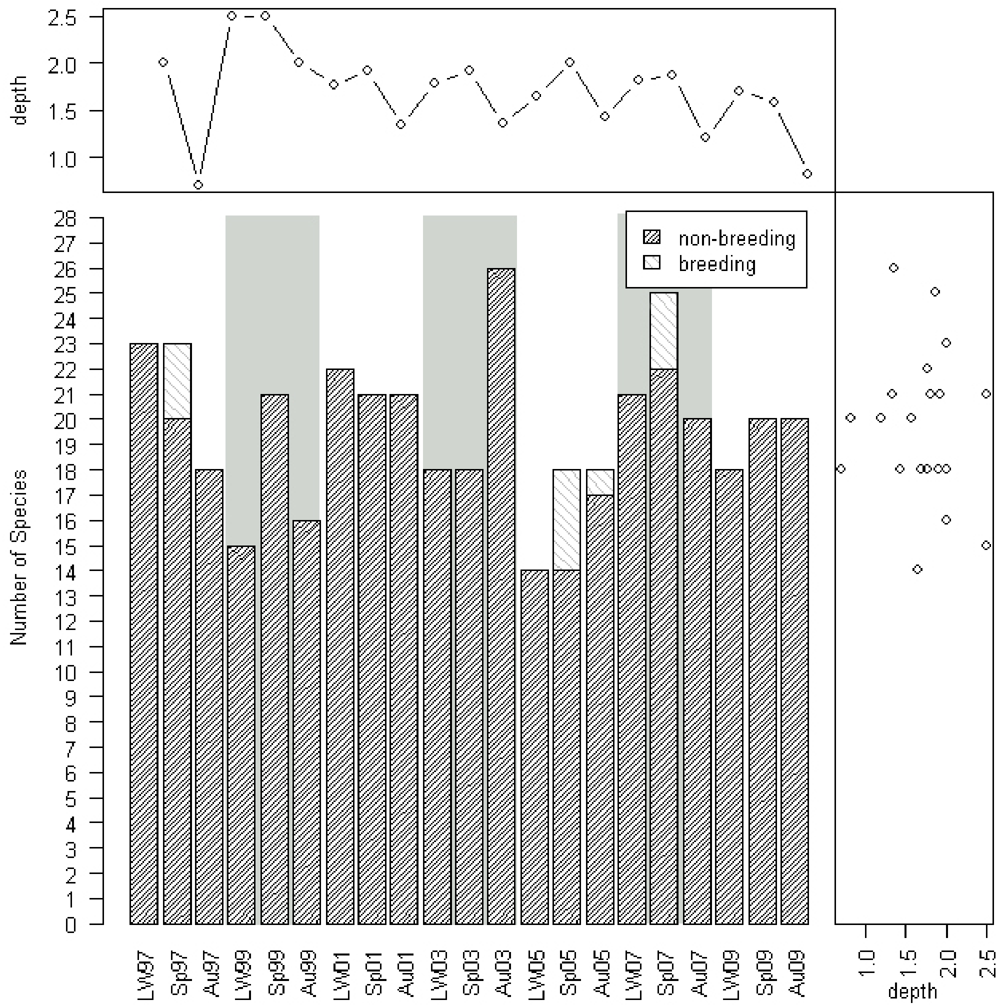

Figure 7. Waterbird species richness divided into breeding and non breeding species for each sampling occasion. Top panel is water depth by sampling occasion. Right panel shows total richness by water depth. Note discontinuous category (sample date) axis.

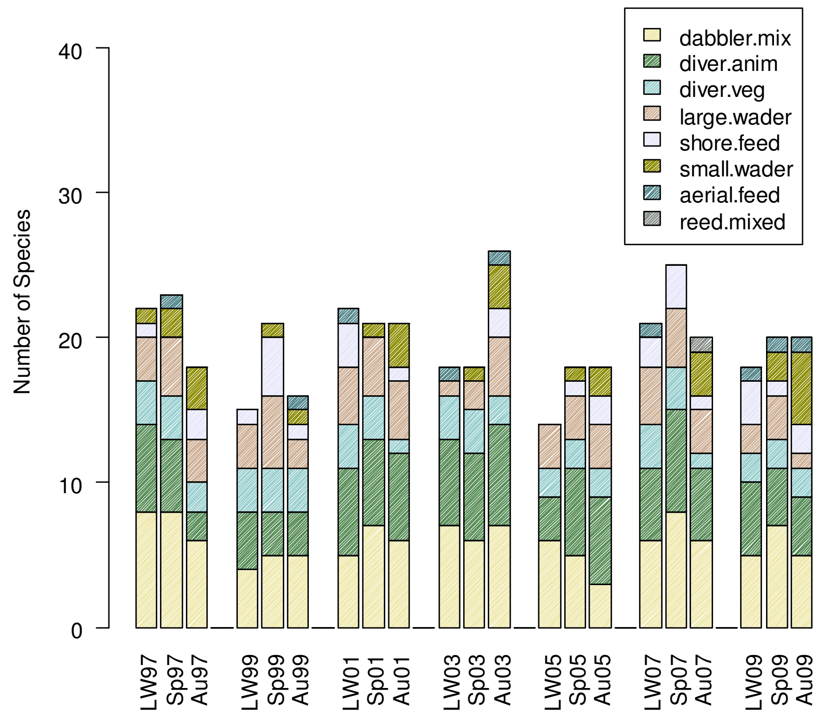

Figure 8. Waterbird species richness distributed amongst feeding guilds for each sampling occasion at Lake Wheatfield.

Temporal changes in the waterbird assemblage

Using an ordination (SSH MDS) of annual species lists and linking successive sampling years graphically, the magnitude and direction of the temporal change in waterbird community composition can be seen (Fig. 9). There is a disjunction between historical and contemporary datasets. Community structure between these periods differs most significantly in species richness. Individual surveys between 1982 and 1985 had a mean richness of 10 species (21 species annually) while between 1997 and 2009 mean richness per survey was 20 species (29 species annually). The historical dataset used in the ordination included two species (Fairy Tern and Red-capped Plover) that were not recorded in the current study. Conversely, data for the current study used in the ordination included 6 species (Yellow-billed Spoonbill, Australian White Ibis, Straw-necked Ibis, Silver Gull, Red-necked Avocet and Black-fronted Dotterel) not recorded in the historical data. Some additional species were recorded only once across both datasets and were therefore excluded from the ordination. These include four species recorded only in the historical dataset (Clamorous Reed-warbler, Australasian Grebe, Marsh Sandpiper and Banded Stilt) and nine species recorded only in the current monitoring program (Banded Lapwing, Spotless Crake, Pied Cormorant, Glossy Ibis, White Sea Eagle, Red-necked Stint, Black-winged Stilt, Red-kneed Dotterel and White-necked Heron).

Within either historical or contemporary dataset a looping progression from year to year as species occur, disappear and re-occur, indicates there is not a sustained directional change in community structure. Changes between years were small to moderate, for example the change between 2001 and 2003 results from 5 species which are not shared between these years (i.e.

15.6% of the two sample species pool). The change between 1997 and 1999 is the largest and the result of 9 species (28% of the two sample species pool) which are not shared.

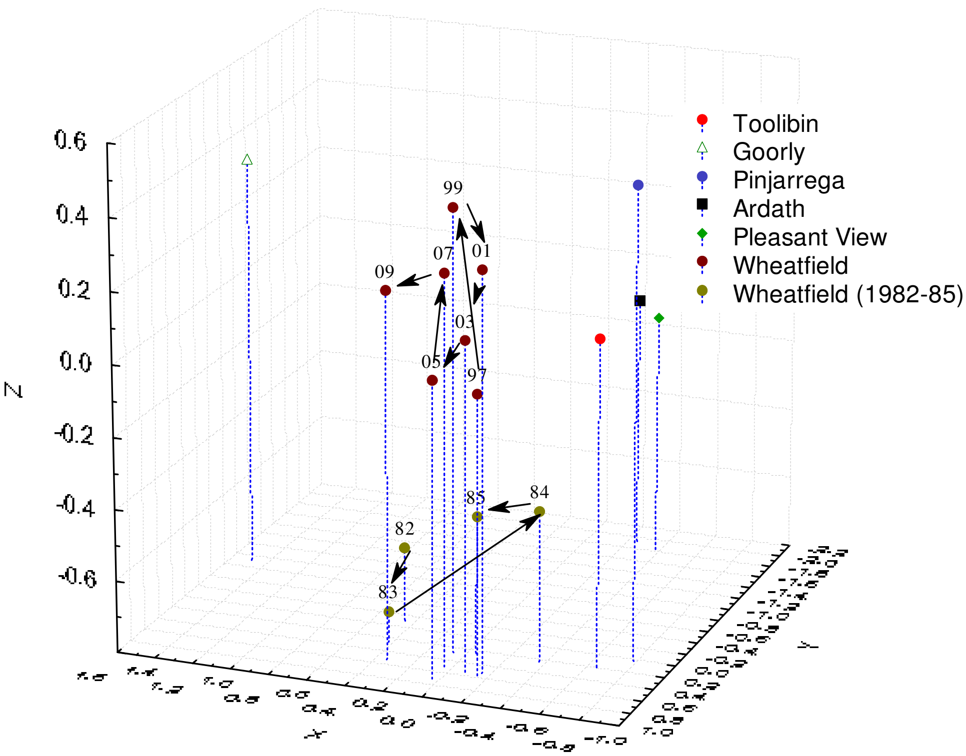

Figure 9. Ordination (SSH Multi dimensional scaling) of Lake Wheatfield waterbird species presence/absence within a survey year, with five "marker" wetlands. Historical data from surveys reported by Jaensch et al (1988). Ordination Stress = 0.11.

Factors associated with change in waterbird assemblage

A classification (UPGMA) of log abundance from individual surveys (from the present monitoring program) was conducted after removing species collected only once. Five groups of surveys were recognised from this classification and are used to colour code samples in an ordination (SSH) of the individual, seasonal waterbird surveys (Fig. 10). Depth, pH, EC and temperature were the only physical variables with significant 'correlation' (MCAO) across this ordination. Vectors (PCC) for these variables were superimposed on the ordination (Fig. 10) and indicate the direction of increase for each variable from the centre of the ordination.

The most meaningful split amongst the individual surveys was between Autumn and Late-winter/Spring samples. Groups 4 and 5 are closely aligned; with the Autumn 2009 survey forced into a separate group by the *a priori* decision that the number of groups would be the square root of the number of surveys. Groups 4 and 5 was composed of 5 of the 7 autumn surveys and occurred during periods of highest EC (group 4 mean = 17.6 mS/cm) and lowest depth (group 4 mean = 1.15 m) and while pH varied it was generally more alkaline than for other classification groups (group 4 mean = 8.71). Species composition within these two groups was characterised by high abundances of most species present, including highest abundances of small waders such as the Common Greenshank and Black-fronted Dotterel (see also Fig. 8). In contrast, Group 1-3 samples occurred at greater lake depths in Late Winter and Spring (Group 1 mean = 1.82 m; Group 3 mean = 1.95 m) and had lower pH (Group 1 mean =

8.15; Group 3 mean = 8.34). Waders were rarely encountered in group 1 to 3 communities and, where present, were less abundant than Group 4 and 5 communities.

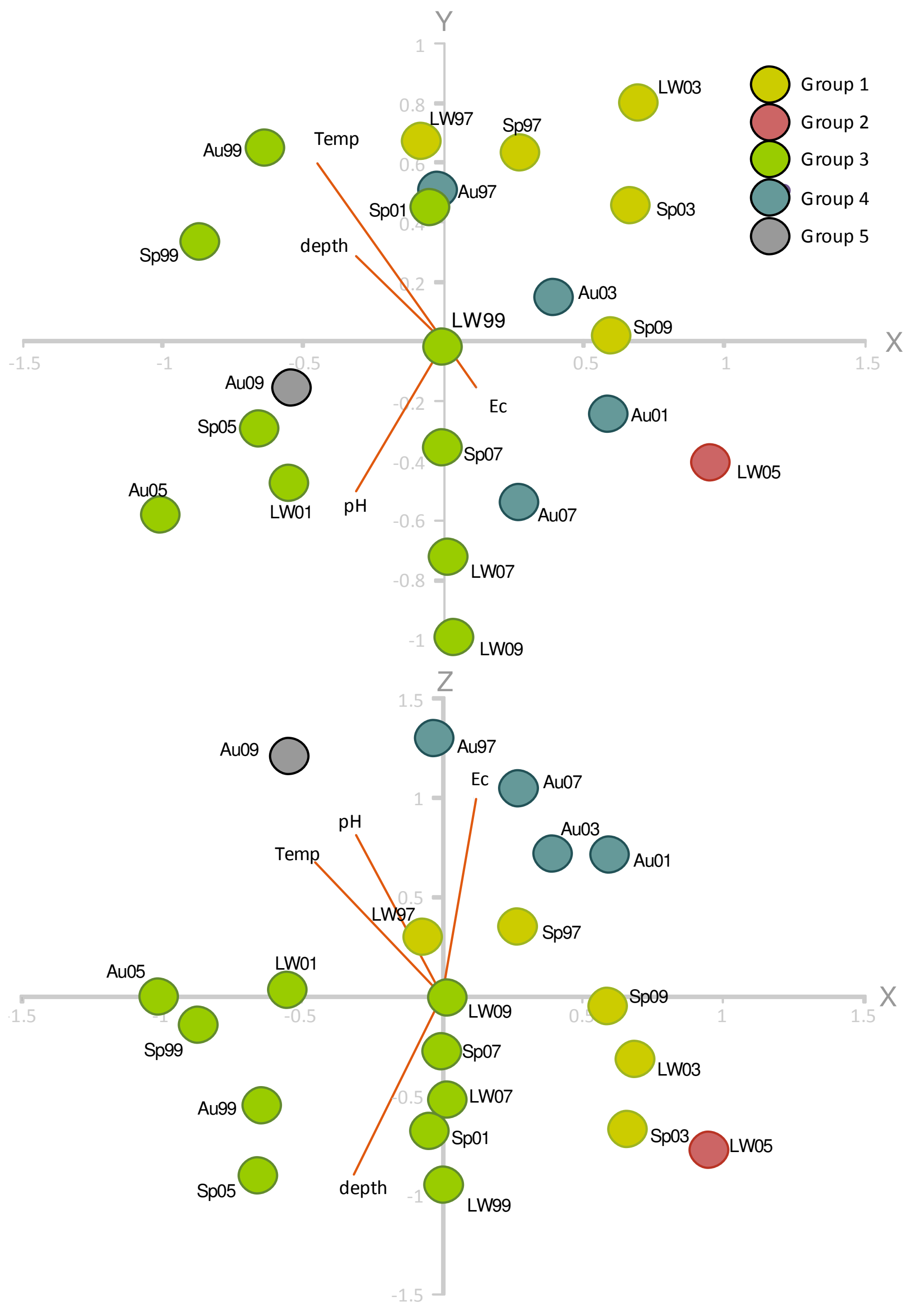

Figure 10. Axes 1 and 2 [top] and axes 1 and 3 [bottom], of a 3 dimensional SSH-MDS ordination of Lake Wheatfield waterbird log abundance, from individual surveys. PCC vectors with MCAO scores <5% are displayed as red lines. Ordination stress = 0.175.

Table 4. Waterbird species list and abundances for Lake Wheatfield from three surveys each sampling year (LW= Late winter. Sp = Spring, Au = Autumn). % Occurrence is the proportion of surveys for which the species was recorded. Figures in parentheses are rank score for the ten most frequently recorded species of Jaensch et al. *(1988)*

	1997			**1999**			**2001**			**2003**			**2005**			**2007**			**2009**			**% Occ.**
Species	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	
Chestnut Teal	429	320	17	55	76	40	96	59	8	39	21	43	12	25	134	8	16	8	46	42	70	100
Eurasian Coot (6)	350	216	226	7	6	2	51	11	31	10	2	227	36	10	31	13	28	81	21	32	54	100
Little Pied Cormorant (7)	15	10	5	60	47	6	7	4	7	5	3	8	2	1	1	7	8	15	6	1	7	100
Pacific Black Duck (3)	78	92	250	55	57	18	71	12	144	5	6	356	10	25	561	32	14	196	18	33	17	100
Grey Teal (1)	51	258	1450	4	82	10	8	20	242	27	31	285	1	2		21	26	269	56	127	144	95
White-faced Heron (4)	2	5	5	3	15	4	4	2	4	8	6	9	1	3	25	1	8	6		2	4	95
Little Black Cormorant	24	79		115	350	272	14	27	129	62	62	84	139	48	268	12	69		5	29	2	90
Musk Duck (8)	37	23	120	6	13	32	25	11		14	6	86	2	7	6	5	5		7	9	4	90
Darter	5			4		4	8	14	19	7	3	18	5	2	4	5	3	6	11	5	6	86
Great Egret	25	4	10	2	30		4	4	4		1	1	1	2	17	2	3	10	3	3		86
Yellow-billed Spoonbill	13	2	30	5	9	24	18	3	57			37	9	20	50	5	10	32	36	6		86
Hardhead	120	16		12	1		1	2		137	8	96	281			2	15	1	15	44		71
Australian White Ibis			1	2	12	5	3		11			11			16	6	7	5	1	1	1	67
Hoary-headed Grebe	14	47	301				6		126	70	31	106			3	8	7	39	2	16		67
Pink-eared Duck	6	84	1670					2	62	2	3	181	2				2	50	7	5	329	67

Table 4. Waterbird species list and abundances for Lake Wheatfield from three surveys each sampling year (LW= Late winter. Sp = Spring, Au = Autumn). % Occurrence is the proportion of surveys for which the species was recorded. Figures in parentheses are rank score for the ten most frequently recorded species of Jaensch et al. *(1988)*

	1997			**1999**			**2001**			**2003**			**2005**			**2007**			**2009**			**% Occ.**
Species	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	
Australian Pelican							3	7	3		1	4		3	5	2	12	30	1		13	57
Blue-billed Duck	2	8		4	7	2	5	11		95	2					11	4					52
Australasian Shoveler	60	3	430			2	5		5	6			1					1			51	48
Australian Shelduck (2)	4	5			5			13	3			4		6	2		3			14		48
Common Sandpiper		2	2		3	3		1	1		1			1						2	17	48
Great Cormorant	1	2		3			59	4	2	3		2		5			1					48
Great Crested Grebe	1	5			2			1			2	2		4	1		2	1				48
Straw-necked Ibis			2		5		250					2		59	22	39	67		486			43
Common Greenshank		1	15						15			20			1			3		1	4	38
Freckled Duck	3	1	3					2			2			1		1	1					38
Black Swan (5)	1					1				2		1				2	2			2		33
Nankeen Night Heron		1			2			2	19			37				1	1					33
Swamp Harrier						1	1			1		1				1			1	1		33
Black-fronted Dotterel	2		5									10			14			3			36	29

Table 4. Waterbird species list and abundances for Lake Wheatfield from three surveys each sampling year (LW= Late winter. Sp = Spring, Au = Autumn). % Occurrence is the proportion of surveys for which the species was recorded. Figures in parentheses are rank score for the ten most frequently recorded species of Jaensch et al. *(1988)*

	1997			**1999**			**2001**			**2003**			**2005**			**2007**			**2009**			**% Occ.**
Species	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	**LW**	**Sp**	**Au**	
Silver Gull (10)					1		10										4		16		1	24
Red-necked Avocet																		6			1	10
Australian Wood Duck	3																					5
Banded Lapwing					2																	5
Black-winged Stilt (9)									3													5
Glossy Ibis					1																	5
Pied Cormorant										2												5
Red-kneed Dotterel												1										5
Red-necked Stint																					1	5
Spotless Crake																		2				5
Whiskered Tern		62																				5
White-bellied Sea-eagle																					2	5
White-necked Heron							1															5
Depth	1	2	0.7	2.5	2.5	2	1.77	1.92	1.34	1.78	1.91	1.36	1.65	2.01	1.43	1.81	1.87	1.2	1.7	1.58	0.82	

Invertebrates

Description of invertebrate assemblage

At least 133 taxa were collected over the monitoring period (Table 6). Annual species richness varied from 42 to 69 (mean = 55.4, sd = 8.4), was not significantly correlated with water chemistry parameters ($-0.55 < r < 0.55$, df = 6, $p > 0.05$) and did not show a consistent increasing or decreasing trend over the monitoring period. Species richness in 2009 (42) was the lowest recorded during the project and 68% of the richness in 1999 (the highest richness). The richness of beetles and non-chironomid dipterans was especially low in 2009. This lower richness coincides with lowest lake depth measured during invertebrate sampling (1.6m).

The invertebrate fauna included species from all 10 of the assemblages defined by Pinder *et al.* (2004). However, several assemblages had low richness and frequency of occurrence (Fig. 12). Assemblage B (typical of claypans and rock pools) was represented by the aquatic beetle *Paroster niger* which was collected only in 1999. Similarly, assemblage C (typical of northern wheatbelt freshwater swamps and lakes) was only recorded in 1999 when three species, *Ecnomus pansus/turgidis* (Trichoptera), *Dero digitata* (Oligochaeta) and *Cladotanytarsus* sp. A (SAP) (Chironomidae) were collected. The only species of Assemblage G (typically crustaceans and insects of naturally saline wetlands) was the dipteran larvae, *Muscidae* sp. A, which was collected only in 2003. Assemblage A (typical of freshwater swamps of the Jarrah forest and Esperance Sandplain) was also represented by only one species; *Dicrotendipes* sp. A V47, but this species was collected in all years except 2009. The remaining assemblages were more frequently collected and represented by 3 to 39 species. Assemblages E (ubiquitous species tolerant of brackish conditions) with 39 species and I (typically encountered in saline streams flowing to the south coast) with 15 species were best represented. Fifty six species (50% of the fauna) could not be placed into an assemblage. Many of these were collected by Pinder *et al.* (2004) but were not included in their assemblage analyses because they were singletons.

Forty percent of species were collected only once during the entire project; i.e. 53 species were singletons. The number of singletons has not declined during the project; some have been re-collected but other new species have been added. The number of new species collected during a monitoring year has declined (49% of annual richness in 1999 to 5% in 2009). This suggests that the bulk of the Lake Wheatfield species pool has now been recorded. However, incidence based estimates of species richness (Chao 2005) have not levelled off after 7 sampling events and suggest the total species pool is in excess of 200 species (Fig. 11).

While singletons comprised a high proportion of the invertebrate community, there was also a substantial core of species which were collected on all, or nearly all, occasions. Most of these species, such as *Austrochiltonia subtenuis* (Amphipoda), *Micronecta robusta* (Hemiptera) and *Chironomus occidentalis* (Chironomidae), are commonly encountered elsewhere. However, a smaller number of species are rarely encountered in the agricultural zone, such as *Cordylophora* sp. (Bryozoa), *Leptocythere lacustris* (Ostracoda) and *Exosphaeroma* sp. (Isopoda).The 23 species in this core group characterise the fauna of the wetland and show it to be distinct from the 4 ‘marker’ wetlands (Fig. 13).

The invertebrate community over time

Linking successive sampling events graphically, following ordination, gives an indication of the magnitude and direction of the change in community structure over time (Fig. 13). The trajectory for community structure has moved backwards and forwards across the ordination space as species have occurred, disappeared and re-occurred. There is no evidence of a directional change over time. A similar lack of progressive change was observed amongst the many water chemistry parameters (e.g. Fig. 5).

Spring pH was the only measured water parameter resulting in a statistically significant PCC vector across an ordination of species presence/absence (Fig. 14). However, vectors for pH and alkalinity (which were correlated), EC and the maximum concentration of phosphate were roughly aligned with grouping of samples derived from a cluster analysis (UPGMA) (Fig. 14). The community from 1997 (group 1) was collected following the highest recorded Late-winter EC, although EC was moderate at the time of the Spring collection. Invertebrate communities sampled in 2003 and 2005 (group 3) formed a 'low salinity and pH' group. These years had the lowest EC and the lowest Late-winter and Spring pH and alkalinity (see also Fig. 5) and community structure included more insect species and fewer rotifers than other groups. The majority of samples (group 2) form a 'moderate salinity and pH' group collected under intermediate values of EC, pH and alkalinity. Community structure in this group had a tendency to include a higher proportion of microinvertebrates than other groups which were more dominated by macroinvertebrates (mainly insects). The annual maxima for total phosphorus concentration during sampling of group 3 ('low salinity and pH') and group 1 (1997) were lower (5-30 mg/L) than for group 2 (30-700 mg/L).

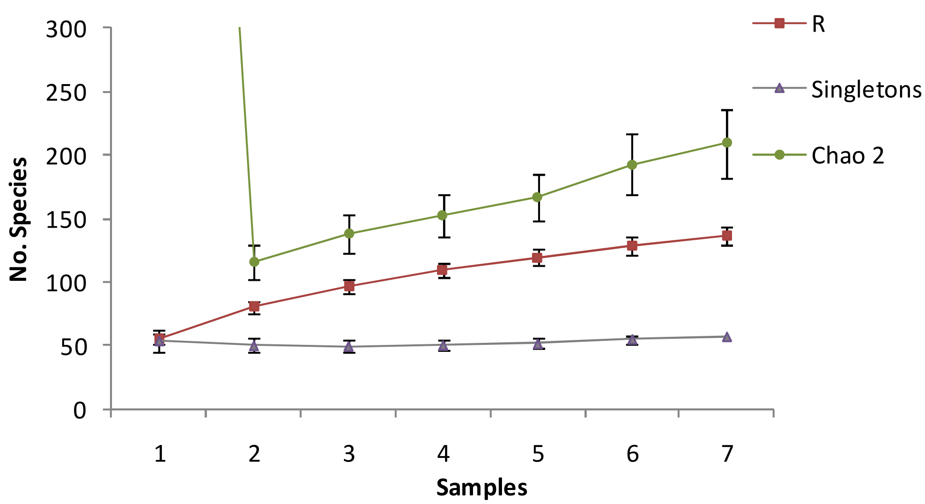

Figure 11. Species accumulation curves for Lake Wheatfield. R= observed species richness, Singletons species collected only once, Chao 2 is the 'classic' estimated species richness based on species incidence. All values are mean +standard deviation.

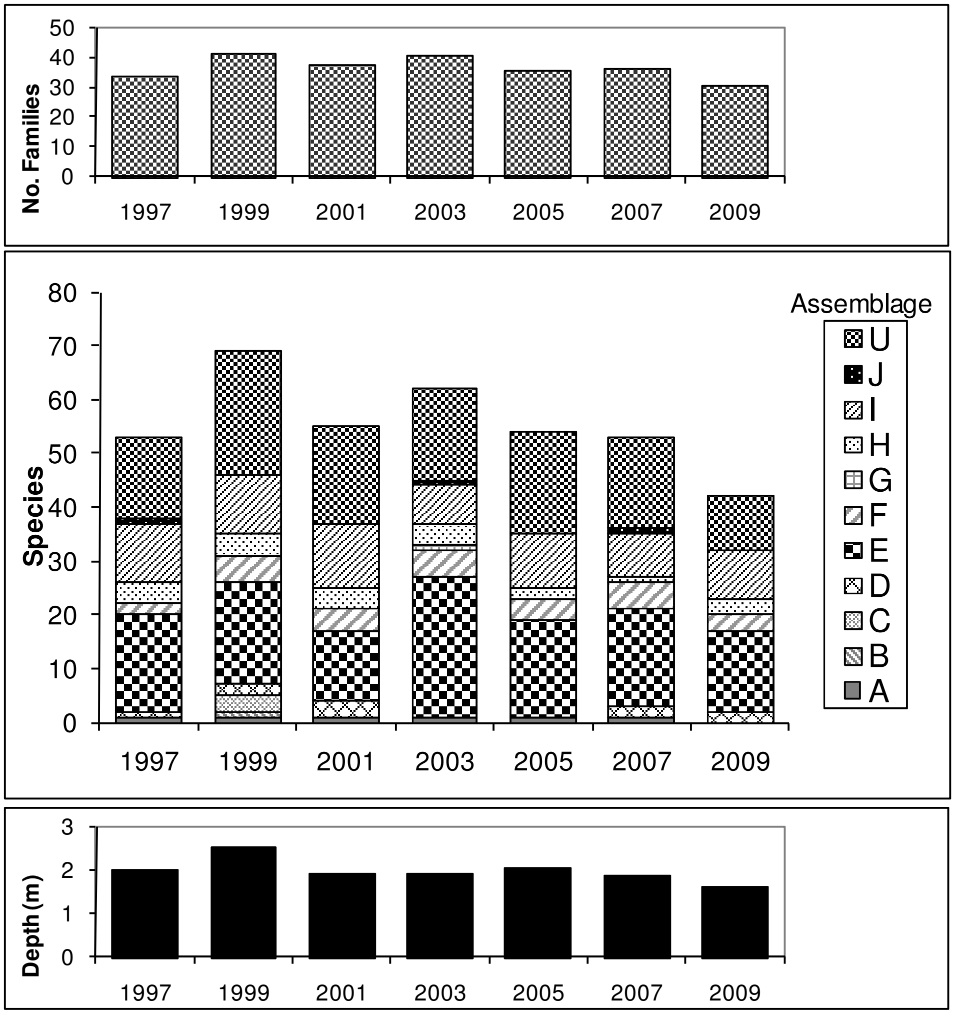

Figure 12. Invertebrate species richness for the seven sampling years at Lake Wheatfield. a) family richness, b) Richness of species within species assemblages of Pinder et al. (2004), c) depth at gauge during sampling.

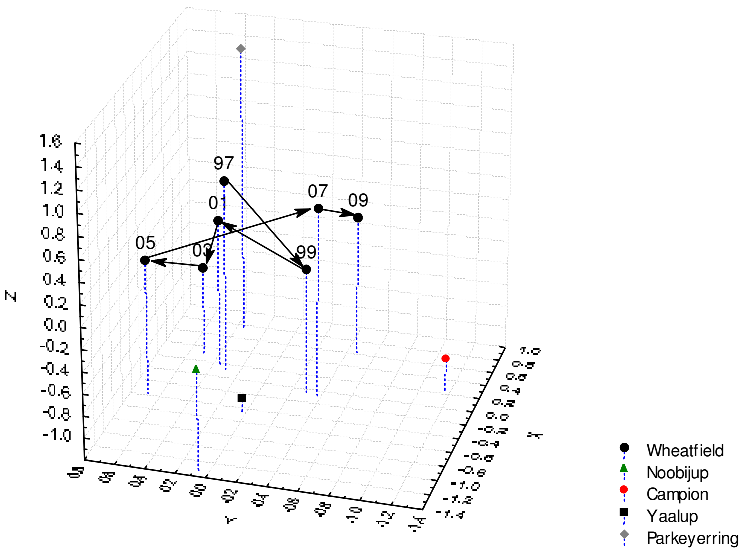

Figure 13. SSH ordination of invertebrate species presence/absence at Lake Wheatfield and the four "marker' wetlands. Arrows link consecutive sampling events

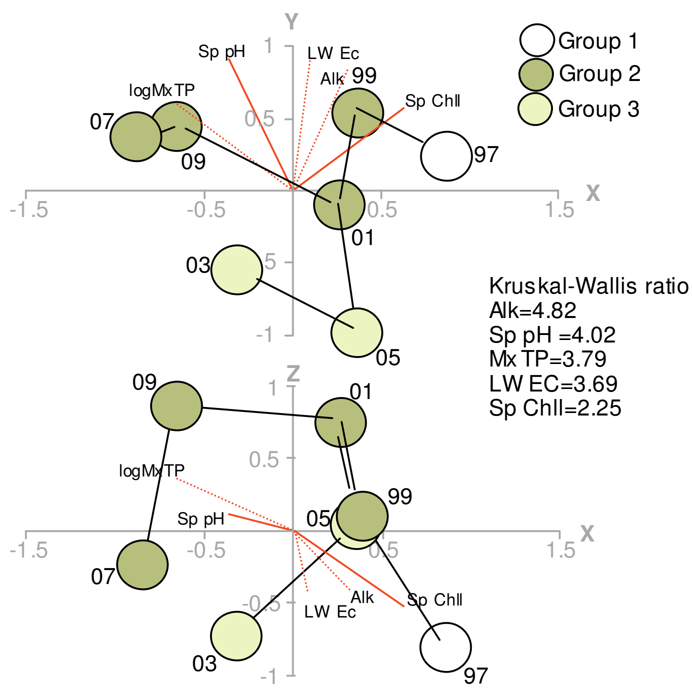

Figure 14. Axes 1 and 2 [top] and axes 1 and 3 [bottom], of a 3 dimension ordination (SSH) of Lake Wheatfield invertebrate presence/absence. Samples are linked by solid lines according to MST. PCC vectors with MCAO scores <5% are displayed as solid lines; broken lines are vectors with best discrimination (Kruskal-Wallis ratio) between groups. Ordination stress = 0.09.

Table 5. Lake Wheatfield invertebrate species list. Assemblages are from Pinder et al. (2004). Values are counts of occurrence in subsites.

LowestID	Assem.	LowestIDNC	1997	1999	2001	2003	2005	2007	2009
HYDRAZOA									
Cordylophora sp.	U	IB020199	2	2	2		1	1	1
TURELLARIA									
Turbellaria (Unident.)	U	IF999999	2				2		
NEMATODA									
Nematoda (unident.)	U	II999999	2	2	2	2	2	2	2
BRYZOA									
Plumatella sp.	U	IO020199						1	
ROTIFERA									
Macrotrachela sp. a (SAP)	U	JB0406A0	1						
Philodinidae	U	JB049999		2					
Bdelloidea	U	JB049999			2			2	2
Hexarthra fennica	H	JF040105	2	2		2			1
Testudinella patina	F	JF050201		1	2			1	1
Brachionus leydigii	U	JP020215		1					
Brachionus plicatilis s.l.	H	JP020219		1	2	1		2	1
Brachionus quadridentatus cluniorbicularis	D	JP020227			1			2	1
Brachionus rotundiformis	I	JP020228	2	1					
Brachionus cf. nilsoni (SAP)	U	JP0202A6			1				
Keratella procurva	F	JP020308		1				2	
Keratella quadrata	U	JP020309		2					
Colurella adriatica	E	JP030101		2		1	1		
Colurella coluris	J	JP030102						1	
Lecane ludwigii	E	JP090136		2			1	1	
Lecane sp. s.str.	U	JP090199		1					
Synchaeta sp.	U	JP150399		1					
Trichocerca sp.	U	JP160399		1					
GASTROPODA									

Table 5. Lake Wheatfield invertebrate species list. Assemblages are from Pinder et al. (2004). Values are counts of occurrence in subsites.

LowestID	Assem.	LowestIDNC	1997	1999	2001	2003	2005	2007	2009
Ascorhis occidua	I	KG021201			2	1	1	2	2
Coxiella sp.	U	KG130299	2	2	1				2
OLIGOCHAETA									
Hirudinea sp.	U	LH999999						1	
Naididae (ex Tubificidae)	U	LO049999		1	1	1	1		
Dero digitata	C	LO050201		2					
Pristina leidyi	U	LO050507				1		1	
Paranais litoralis	I	LO050801		1	1	1	1	2	2
Enchytraeidae	U	LO089999		2	2	2	2	1	1
ARANAEA- *Mites*									
Hydrachnidae	U	MM019999	1						
Koenikea nr australica (=verrucosa)	U	MM169999		1			1		
Acercella falcipes	E	MM170101				2			
Pezidae	U	MM259999		1					
Oribatida	U	MM9999A1	1	2	2	2	2	1	1
Mesostigmata	U	MM9999A2	2	2	2	1	2	1	
Trombidioidea	U	MM9999A6		1		1	2		
CLADOCERA									
Alona sp.	U	OG030299				1			
Pleuroxus inermis	E	OG032502				1			
Daphnia carinata	E	OG040201	1			2	2		2
Macrothrix breviseta	E	OG060201			2	2		2	1
OSTRACODA									
Cyprideis australiensis	I	OH040101	1	2	1	2	2	2	1
Cytherideidae	U	OH049999					2		
Ilyocypris australiensis	E	OH060101					1		
Alboa worooa	F	OH080101	2		1	2	2		
Bennelongia sp.	U	OH080399					1		
Candonocypris novaezelandiae	U	OH080403						2	2

Table 5. Lake Wheatfield invertebrate species list. Assemblages are from Pinder et al. (2004). Values are counts of occurrence in subsites.

LowestID	Assem.	LowestIDNC	1997	1999	2001	2003	2005	2007	2009
Diacypris spinosa	H	OH080703	2	1	1		1		1
Mytilocypris mytiloides	U	OH081204	2			2	1	1	
Reticypris clava	H	OH081501	2	1	1				
Ilyodromus sp.	U	OH081999							1
Zonocypris sp BOS082	U	OH0828A1					2		
Sarscypridopsis aculeata	E	OH090101	2	2	2	1	2	2	2
Leptocythere lacustris	I	OH100101	1	2	2	2		2	2
Kennethia cristata	D	OH110201	1	1	1			1	
COPEPODA									
Gladioferens imparipens	I	OJ110401	2	2	2	2	2	2	2
Metacyclops sp. 442 (CB)	E	OJ3102A0							1
Halicyclops sp. 1 (nr ambiguus) (SAP)	I	OJ3104A0	2	1	1	2	1		2
Mesocyclops brooksi	F	OJ310703		2		1		2	1
Mesochra baylyi	I	OJ610302	1						
Cletocamptus aff deitersi	U	OJ6104A0	1						
Onychocamptus bengalensis	I	OJ620101	2	2	2	2	2	2	2
Schizopera clandestina	J	OJ630201				2			
Nitocra reducta	U	OJ640101		1	2	2	2	1	
Nitocra sp. 4 (SAP)	D	OJ6401A4		1	2				1
AMPHIPODA									
Austrochiltonia subtenuis	E	OP020102	2	2	2	2	2	2	2
Melita kauerti	I	OP090402	2	2	2		2		
ISOPODA									
Exosphaeroma sp.	I	OR130299	2	1	2		1	1	1
DECAPODA									
Palaemonetes australis	I	OT020201	2	1	1	2	2	2	1
COLEOPTERA									
Allodessus bistrigatus	E	QC091101					1		
Paroster niger	B	QC091407		1					

Table 5. Lake Wheatfield invertebrate species list. Assemblages are from Pinder et al. (2004). Values are counts of occurrence in subsites.

LowestID	Assem.	LowestIDNC	1997	1999	2001	2003	2005	2007	2009
Antiporus femoralis	F	QC091604				1			
Sternopriscus multimaculatus	E	QC091805	1	1	1				
Necterosoma sp.	U	QC092099	1						
Lancetes lanceolatus	E	QC092401	1						
Berosus sp.	U	QC110499	1	2		2	1		
Enochrus sp.	U	QC111199						1	
Hydrophilidae	U	QC119999			1				
Ochthebius sp.	U	QC130399			1			1	
Gymnocthebius sp. 1 (SAP)	I	QC1304A0	1						
Limnichidae	U	QC359999			1				
DIPTERA									
Bezzia sp.	E	QD090499					1		
Clinohelea sp.	U	QD090699					1		
Culicoides sp.	E	QD090899	1	1	2	2	1	1	
Monohelea sp. 3 (SAP)	E	QD0919A2		1	1				
Nilobezzia sp.	E	QD092099		2		1	1	2	1
Tabanidae	U	QD239999			1	1	1		
Stratiomyidae	U	QD249999	1	2		1		2	
Empididae	U	QD359999		1					
Dolichopodidae	U	QD369999	1	1					
Sciomyzidae	U	QD459999				1			
Ephydridae sp. 2 (SAP)	E	QD7899A6	1					1	
Ephydridae sp. 3 (SAP)	H	QD7899A7	1			1			
Ephydridae sp. 6 (SAP)	H	QD7899B0			1		1		
Ephydridae sp. 7(SAP)	U	QD7899B1			1				
Muscidae sp. A (SAP)	H	QD8999A0				1			
Muscidae sp. D (SAP)	G	QD8999A3				1			
Procladius paludicola	E	QDAE0803	2	2	2	2		2	2
Procladius villosimanus	E	QDAE0804	2			1	2		

Table 5. Lake Wheatfield invertebrate species list. Assemblages are from Pinder et al. (2004). Values are counts of occurrence in subsites.

LowestID	Assem.	LowestIDNC	1997	1999	2001	2003	2005	2007	2009
Corynoneura sp. (V49) (SAP)	U	QDAF06A2			1	1	2		
Paralimnophyes pullulus	F	QDAF1202		2	1	1	2	1	
Cladotanytarsus sp. A (SAP)	C	QDAH03A0		2					
Tanytarsus fuscithorax/semibarbitarsus	E	QDAH0410	2	2	2	2	2	1	1
Chironomus occidentalis	F	QDAI0408	2	2	2	2	2	2	2
Chironomus aff. alternans (V24) (CB)	E	QDAI04A0	2			2	2		
Dicrotendipes conjunctus	E	QDAI0603	2	2	2	2	2	2	2
Dicrotendipes sp. A (V47) (SAP)	A	QDAI06A0	2	2	2	2	2	2	
Kiefferulus intertinctus	E	QDAI0701		2		1		2	2
Polypedilum nubifer	E	QDAI0804		1					2
Polypedilum nr vespertinus (M2) (SAP)	I	QDAI08A1			1				
Polypedilum nr. convexum (SAP)	J	QDAI08A2	2						
Cryptochironomus griseidorsum	E	QDAI1901		1	1				2
Cladopelma curtivalva	E	QDAI2201	1			1			
HEMIPTERA									
Agraptocorixa eurynome	E	QH650301				2			
Agraptocorixa sp. (juveniles)	U	QH650399						1	
Micronecta robusta	E	QH650502	1	2	1	2	1	1	2
Anisops thienemanni	E	QH670401				1			
Anisops hackeri	U	QH670405		1					
Notonectidae (juveniles)	U	QH679999							1
LEPIDOPTERA									
Lepidoptera (non-pyralid) nr.sp.16 (SAP)	U	QL9999A0			1				
ODONATA									
Austroagrion cyane	U	QO020501					1		
Ischnura heterosticta heterosticta	E	QO021002				1			
Xanthagrion erythroneurum	E	QO021301				1	1		
Austrolestes analis	F	QO050101					2		
Austrolestes annulosus	E	QO050102	2	2	1	2		1	

Table 5. Lake Wheatfield invertebrate species list. Assemblages are from Pinder et al. (2004). Values are counts of occurrence in subsites.

LowestID	Assem.	LowestIDNC	1997	1999	2001	2003	2005	2007	2009
Austrolestes aridus	E	QO050103		2		2		2	
Austrolestes io	E	QO050105					2	1	1
Austrolestes sp.(early instar)	U	QO050199			1				1
Hemianax papuensis	E	QO121201				1		1	
Hemicordulia tau	E	QO300102	1	1		1			
TRICHOPTERA									
Ecnomus pansus/turgidus	C	QT0804A0		2					
Notalina spira	E	QT250504	2	2	1	2	2	2	1
Oecetis sp.	U	QT250799	2	2				2	
Symphitoneuria wheeleri	I	QT250903		2	1		1		
Triplectides australis	E	QT251103	1					2	
Leptoceridae (early instar)	U	QT259999				1			

Groundwater

All five observation bores at Lake Wheatfield showed an oscillation in water level consistent with wet-dry seasonal cycles (Fig. 15). The overall trend in each observation bore was an increase in water level (Australian Height Datum - AHD) over the study period. The average rate of increase in groundwater level across observation bores one to five was 5.3 cm year^{-1} with the greatest increase, 6.48 cm year^{-1}, observed at LW05OB, which is the bore located SSE of the lake and closest to transect 3. The annual increase per bore was determined using the slope value of each linear fit (m/day) x 365 from which the mean was calculated.

The electrical conductivity of groundwater (ECg) ranged from 77 ms m^{-1} to 4400 ms m^{-1} (Fig. 16). In general, water sampled from LW01OB had a higher ECg than the remaining observation bores. There was no apparent linkage between ECg and groundwater depth. In 2007 LW02OB and LW03OB showed a spike in ECg which was not observed in the remaining bores.

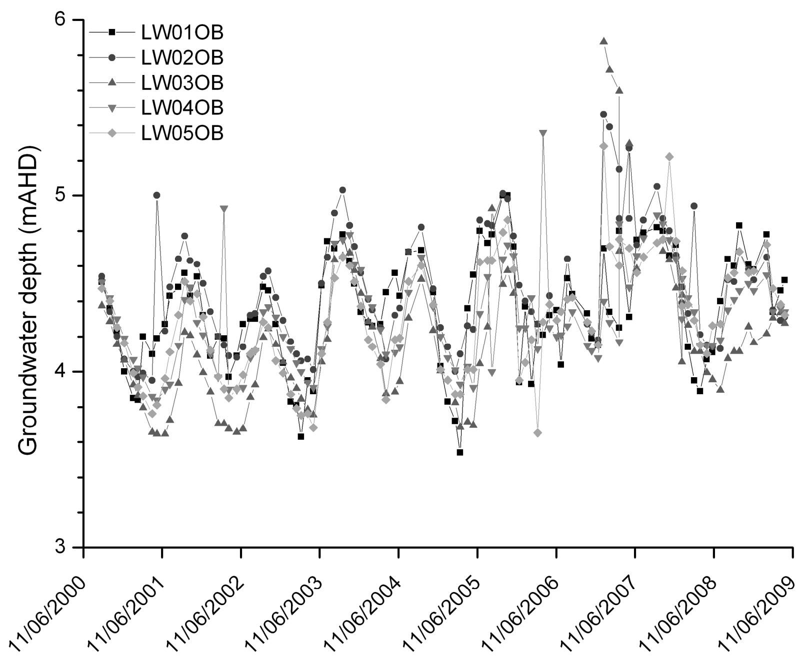

Figure 15. Groundwater level (mAHD) at Lake Wheatfield at observation bores 1-5.

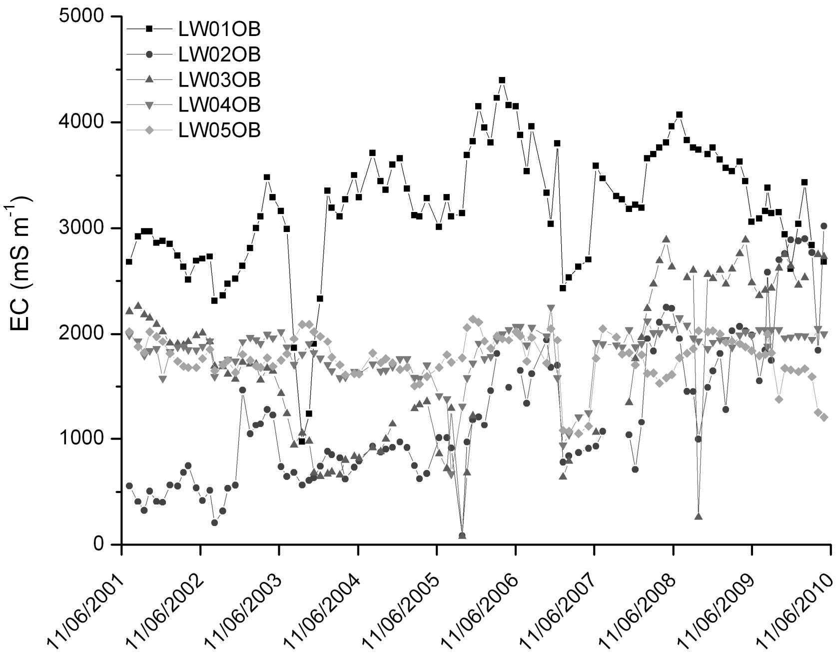

Figure 16. Groundwater electrical conductivity (EC) at Lake Wheatfield at Observation Bores 1 to 5.

Vegetation transects

Soil electrical conductivity

Soil electrical conductivity (EC_a) showed a similar spatial pattern at transects 1, 2 and 4 (Fig. 17). Within these transects EC_a values were closely related to elevation. Soil electrical conductivity was consistently lower at the higher elevations associated with the upland section of transects and sandy rises. Maximum EC_a values were recorded beside channels (beginning of transect 1 Fig. 17A), and within the swales and at the terminal point of transects which were on the lake floor (Fig. 17c -T2, Fig. 17e - T4).

There was a decrease in EC_a in 2009 in comparison to 2006 at transects 1, 2 and 4 (Fig. 17). Soil electrical conductivity at transect 4 was greatest in 2003. The lowest EC_a values occurred at the highest part of the landscape at transect 2 (Fig. 17) and likewise the highest EC_a values were recorded at transect 3 which was almost completely inundated throughout all monitoring years (data not shown). Only a fraction of transect 3 was not inundated and this section also had lower EC_a values in 2009 than in previous sampling periods.

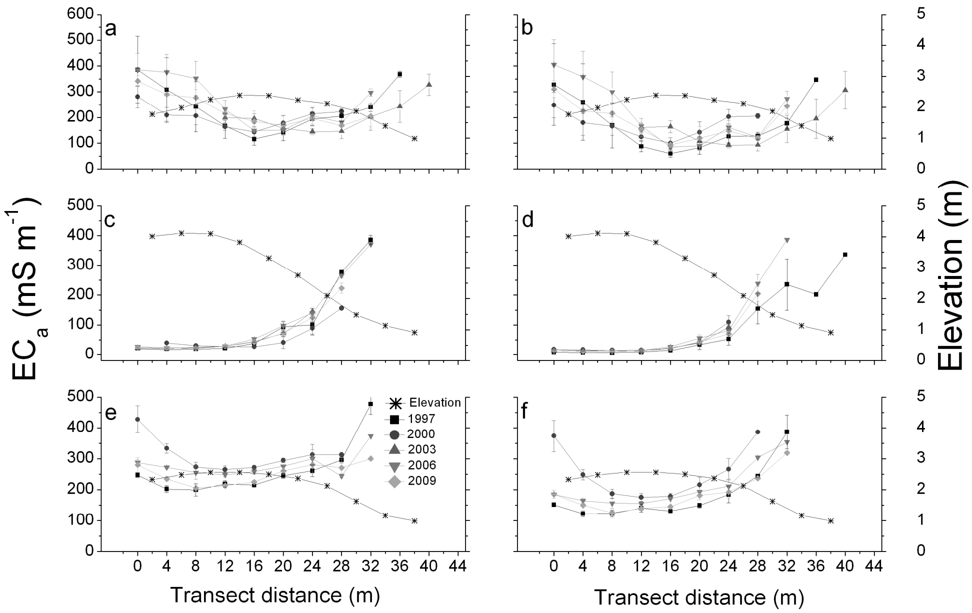

Figure 17. Soil electrical conductivity (EC_a), and transect elevation relative to the lake floor at transects 1 (a & b), 2 (c & d) and 4 (e & f) for Lake Wheatfield between 1997 and 2009. Each value for EC_a is the mean (± s.e.) of three points across the transect. EM38 measurements in vertical dipole (V) orientations are in the left column; horizontal dipole (H) in the right column.

Transect elevation and inundation

Mean elevation, relative to the lake floor, varied within and between transects (Fig. 18). In general there was a slight grade from the terrestrial component of each transect (towards the relative coordinate zero on the transect distance axis), except at transect 2 where the difference in elevation was larger, spanning approximately 3 m. Lake water levels present over the study period meant that transect 3 was almost always completely inundated whereas the remaining transects were only inundated at lower elevations (Fig. 18).

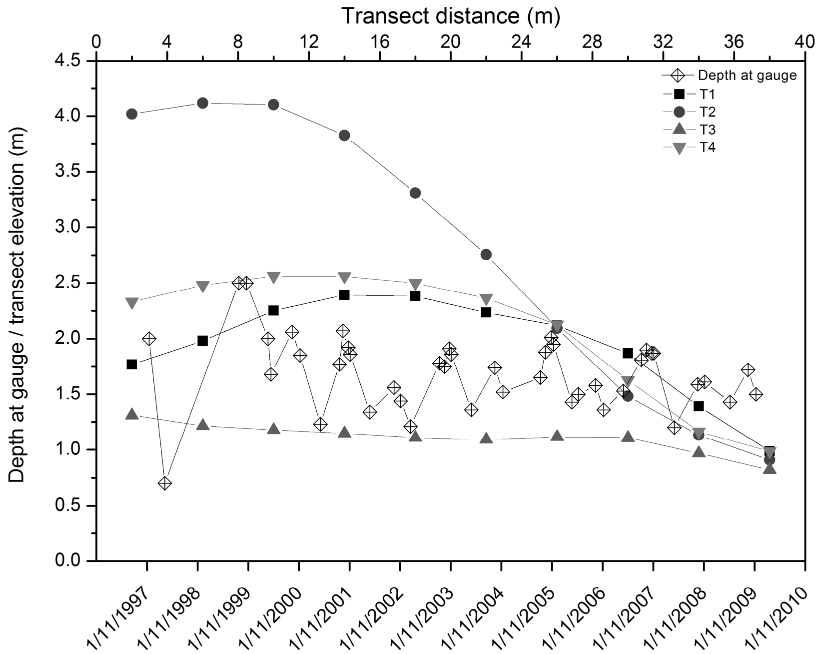

Figure 18. Vegetation monitoring transect (T1-T4) elevations at Lake Wheatfield. The depth of standing water over the monitoring period is also shown. Note pre 2000 water depths are estimates prior to gauge installation.

Vegetation condition

Trends in basal area and population structure

Total basal area (the sum of the basal area from all four transects) declined over the study period (Fig. 19a). This pattern was largely driven by a reduction in basal area within transects 1, 2 and 3 (data for all species at each transect pooled) (Fig. 19a). By contrast, transect 2 showed an increase in basal area after 2003. A prescribed fire (DEC management burn) in spring 2009, just prior to the survey, was responsible for much of the decline in basal area in transect 1. The decline in basal area in transect 3 was due to the death of many *Melaleuca cuticularis* trees (26.5% of the total transect basal area), the only tree species present in this transect. By contrast, basal area within transect 4 was relatively stable resulting from recruitment of *Melaleuca brevifolia* on the elevated section of the transect. Excluding transect 1, which was impacted by fire, the greatest basal area decline was observed in transect 3 with the largest decrease occurring between 2006 and 2009, suggesting that many trees died during this period. *Melaleuca cuticularis* mortality also contributed towards a significant amount of basal area decline at transect 1, but was stable at transect 4 (Fig. 19b).

Transect 2 was dominated by *Banksia speciosa*. A wildfire burnt the entire transect in 2003 and no mature *B. speciosa* was recorded in the 2003 (post fire) measurement period (Fig. 20). A spike in the number of *B. speciosa* in the 0-20 cm DBH (seedling) size class was recorded during the same year. As the post fire stand matured, the number of *B. speciosa* recorded in larger DBH size classes increased and was matched by a decrease in the number of seedlings

For inundated sections of transect 2 the basal area of *Melaleuca cuticularis* declined (Fig. 21a); but in more elevated parts of the transect basal area increased. Overall there was a 22% reduction in basal area of *M. cuticularis* over the study period within the transect and no recruitment was recorded (Fig. 21b).
Basal area of *M. cuticularis* at transect 3 (Fig. 22a) declined at nearly all parts of the transect and across most size classes over the study period.

The basal area at transect 4 of *M. cuticularis*, which occurred at a high elevation, increased during the monitoring period (Fig. 25). Analysis of the population demography at transect 4 provided evidence of recruitment between 2003 and 2006. The basal area of *Melaleuca brevifolia* also increased at transect 4 and the population structure indicates a transition from smaller to larger size classes, consistent with growth in this species (Fig. 26).

At the sub-plot scale (transect sub-plots pooled) change in basal area of *M. cuticularis* between 1997 and 2009 were positively correlated with elevation (Fig. 23). At low elevations (less than 1.45 m from the lowest point of the wetland) the basal area of *M. cuticularis*, declined over the study period whereas at higher elevations basal area was stable or increased slightly (Fig. 24).

There was a general pattern of decline for *Eucalyptus incrassata* at transect 4 (Fig. 27a). The greatest decline occurred between 2006 and 2009 and a substantial decline between 2000 and 2003. The DBH size class distribution is indicative of death in larger trees (Fig. 27b).

Across transects the percentage decline in *M. cuticularis* basal area between 1997 and 2009 was negatively correlated with the average EC_a (vertical dipole) recorded over the study (Fig. 28). This meant that, on average, basal area started to decline where average EC_a was greater than 250 mS m^{-1}.

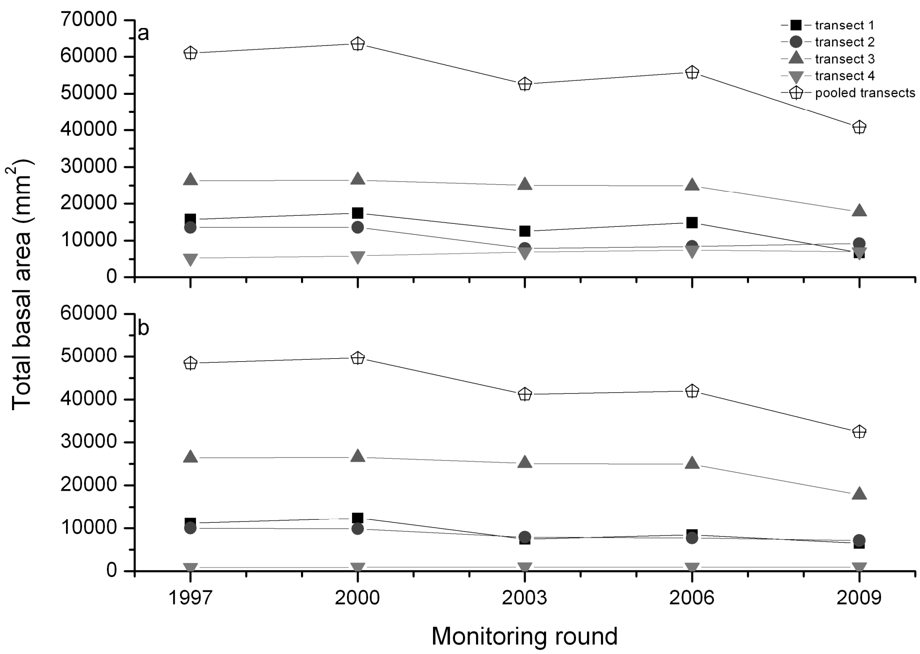

Figure 19. T*otal and transect basal area (mm²) for* ***(a)*** *all overstorey species and* ***(b)*** Melaleuca cuticularis *at Lake Wheatfield.*

Species included in Fig. 19 are *Acacia cyclops, A. saligna, A.* sp.2, *Banksia speciosa, Eucalyptus incrassata, E. occidentalis, Melaleuca brevifolia, M. cuticularis, Nuytsia floribunda* and *Spyridium globulosum.*

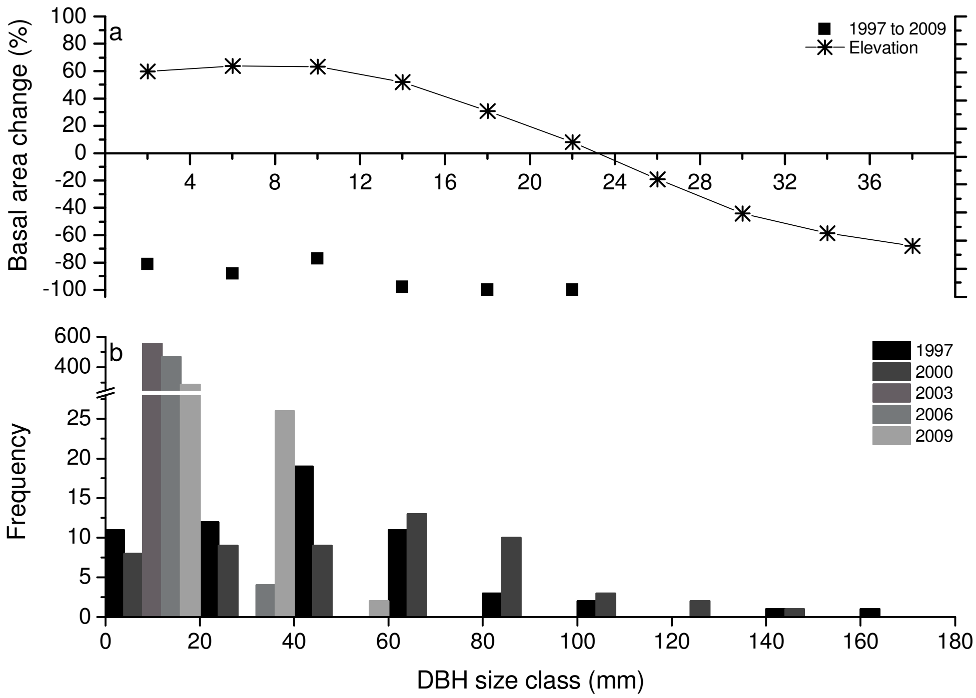

Figure 20. Banksia speciosa *at transect 2. **(a)** Percent change in stand basal area (1997 - 2009) for each 4 x 20m subplot along the transect (solid squares). Higher transect distance is closer to the wetland. **(b)** Frequency histogram of diameter at breast height (DBH) for all sample years.*

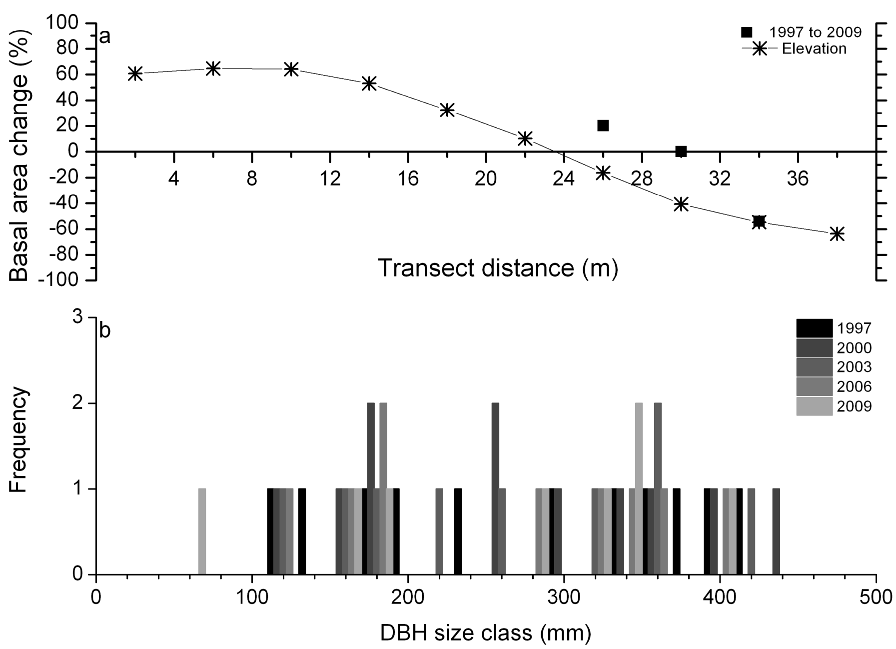

Figure 21. Melaleuca cuticularis *at transect 2. **(a)** Change in stand basal area (1997 -2009) for each 4 x 20m subplot along the transect (solid squares). Higher transect distance is closer to the wetland. **(b)** Frequency histogram of diameter at breast height (DBH) for all sample years.*

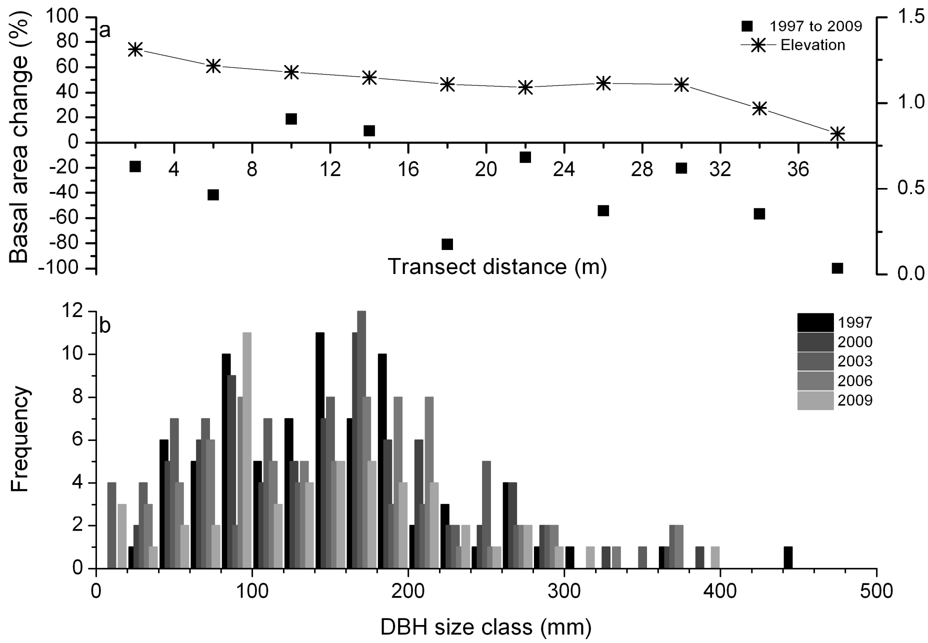

Figure 22. Melaleuca cuticularis *at transect 3. **(a)** Change in stand basal area (1997 -2009) for each 4 x 20m subplot along the transect (solid squares). Higher transect distance is closer to the wetland. **(b)** Frequency histogram of diameter at breast height (DBH) for all sample years.*

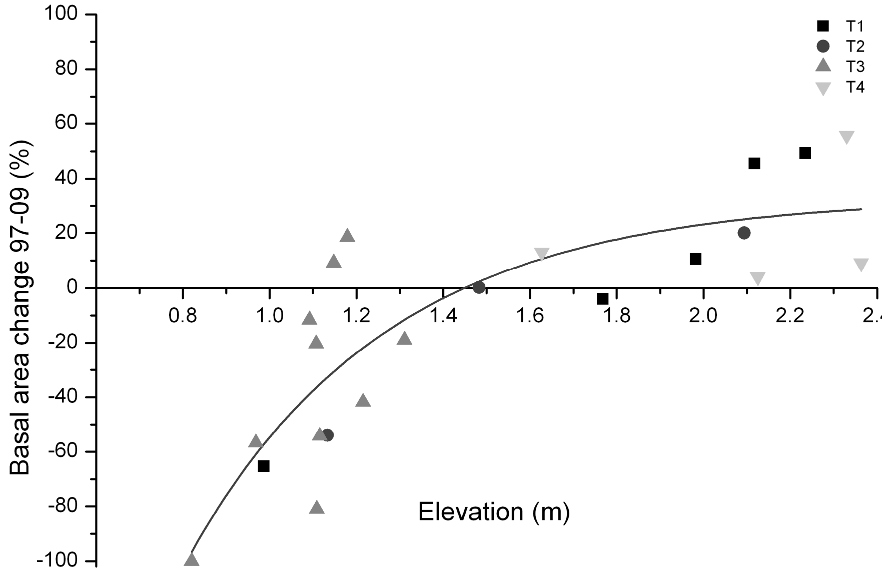

Figure 23. Percentage change in sub-plot basal area for Melaleuca cuticularis *at Lake Wheatfield, plotted against sub-plot mean elevation. Basal area data includes monitoring events from 1997-2009 except for transect 1 where 2009 data is excluded due to fire damage to the transect* R^2 is 0.71, $p < 0.001$

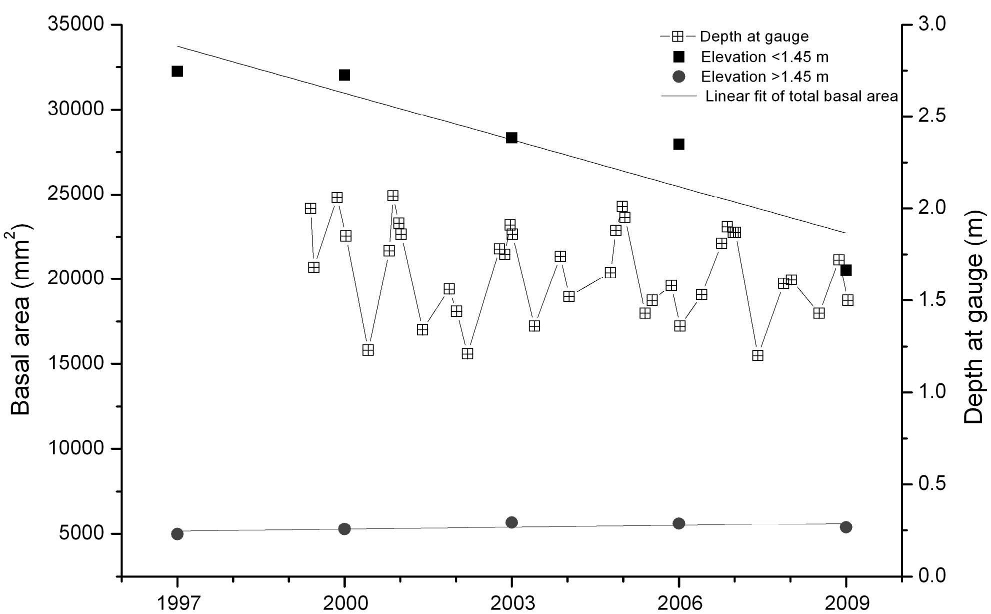

Figure 24. Basal area of Melaleuca cuticularis *at Lake Wheatfield showing decline where flooded since 1999 (< 1.45 m elevation, squares) compared with the same species higher in the profile (>1.45 m elevation, circles). Transect 1 data is not included due to fire in 2009 prior to the survey. <1.4m R^2 = 0.78894, p = 0.02812; >1.4m R^2 = 0.24559, p = 0.22648*

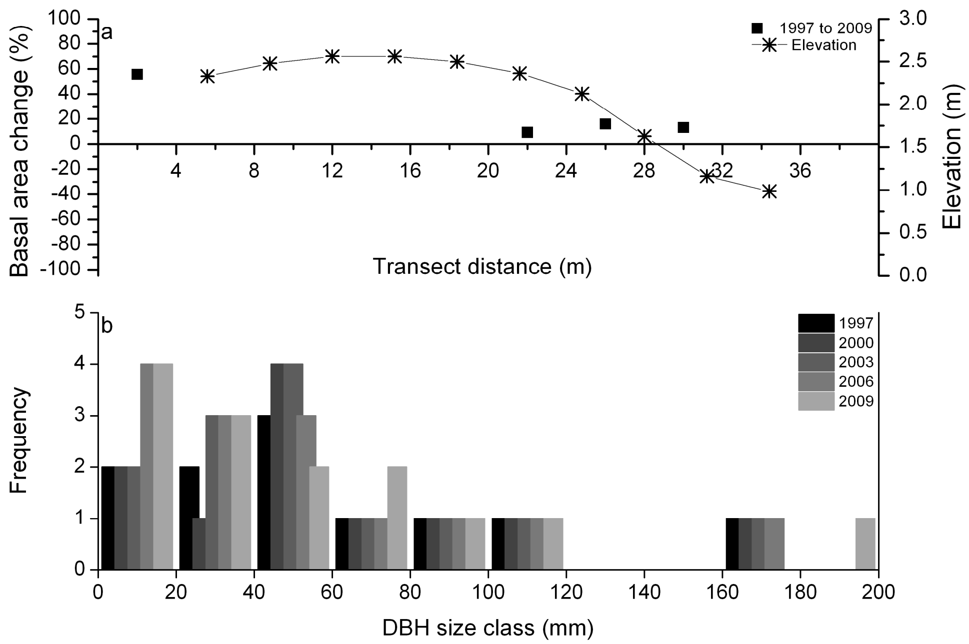

Figure 25. Melaleuca cuticularis *at transect 4.* ***(a)*** *Change in stand basal area (1997 -2009) for each 4 x 20m subplot along the transect (solid squares). Higher transect distance is closer to the wetland.* ***(b)*** *Frequency histogram of diameter at breast height (DBH) for all sample years.*

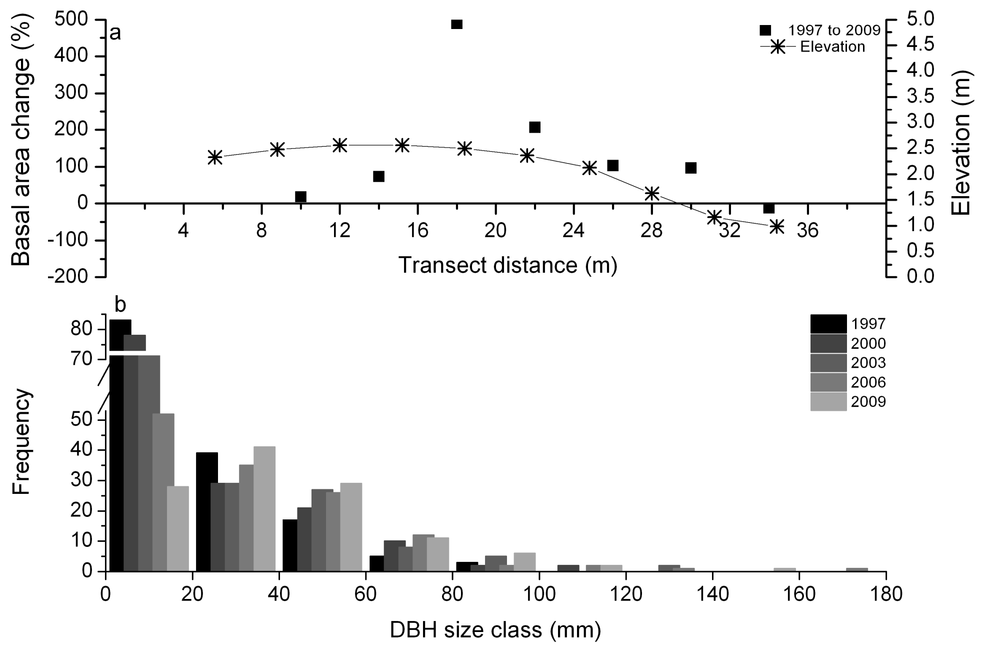

Figure 26. Melaleuca brevifolia *at transect 4* ***(a)*** *Change in stand basal area (1997 -2009) for each 4 x 20m subplot along the transect (solid squares). Higher transect distance is closer to the wetland.* ***(b)*** *Frequency histogram of diameter at breast height (DBH) for all sample years.*

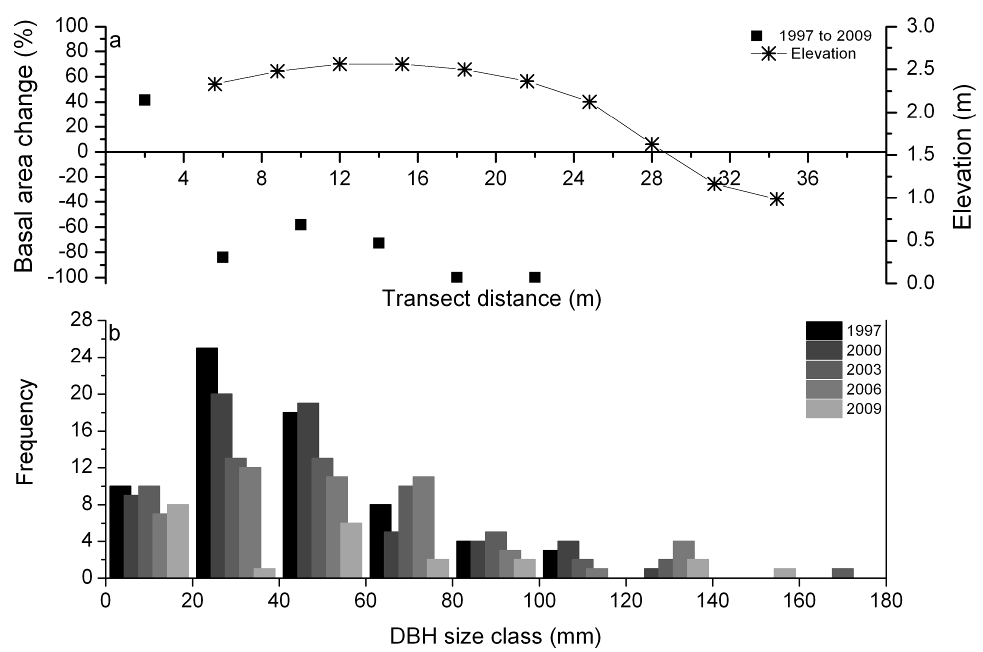

Figure 27. Eucalyptus incrassata *transect 4.* ***(a)*** *Percentage change in stand basal area (1997 -2009) for each 4 x 20m subplot along the transect (solid squares). Higher transect distance*

*is closer to the wetland. **(b)** Frequency histogram of diameter at breast height (DBH) for all sample years.*

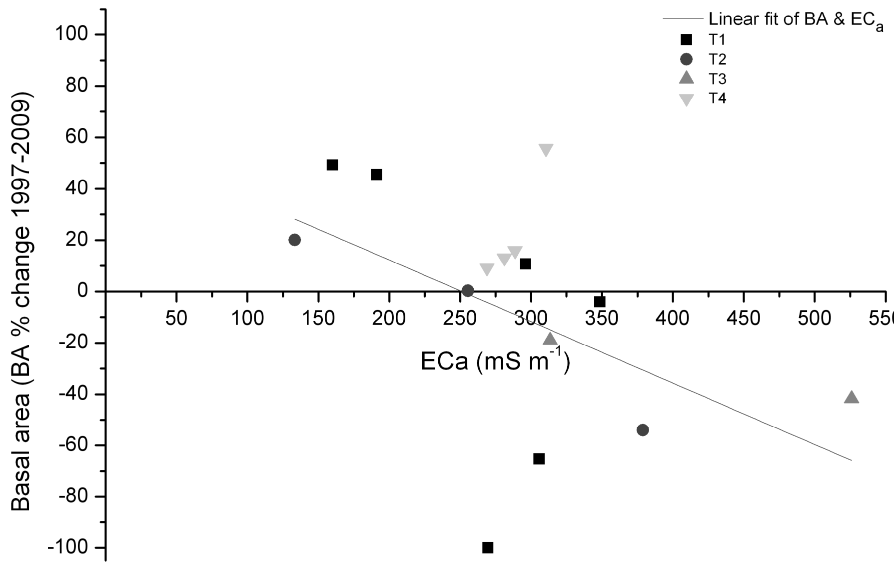

Figure 28. Percentage change in sub-plot basal area (BA) for Melaleuca cuticularis *plotted against sub-plot soil electrical conductivity (EM38 – EC_a – vertical dipole) (average of all years data except T4 in 2003) for transects at Lake Wheatfield. Linear regressions are shown for all transects pooled (R^2 = 0.23333, p = 0.04591).*

Trends in canopy condition

Although there was substantial year-to-year variation in the canopy health of *M. cuticularis*, transect 1 and 3 had the lowest average index of canopy health and transect 4 had the highest (Fig. 29). Friedman non-parametric ANOVA confirmed significant differences ($p<0.05$) between years for all occurrences of *M. cuticularis* for surviving trees and the stand through time (Table 7). The mean canopy health of *M. cuticularis*, whether calculated with dead trees included in all years (Fig. 29b) or with dead trees excluded after the first record of death (Fig. 29a), declined over the study period in all transects except transect 4 where condition improved.

Comparison of health rating of *M. cuticularis* between 1997 and 2009 (Wilcoxon pairwise test) revealed significant declines ($p<0.05$) in condition with the exception of transect 4 (improved) and surviving trees at transect 2 (Table 7).

Eucalyptus incrassata (transect 4) also showed significant differences ($p<0.05$) between years and decline in condition over the duration of monitoring (Table 7). At the same transect *M. brevifolia* showed significant differences between years. Surviving *M. brevifolia* plants showed significantly higher condition scores in 2009 compared to 1997. The stand as a whole showed no significant change over the same period (Table 7).

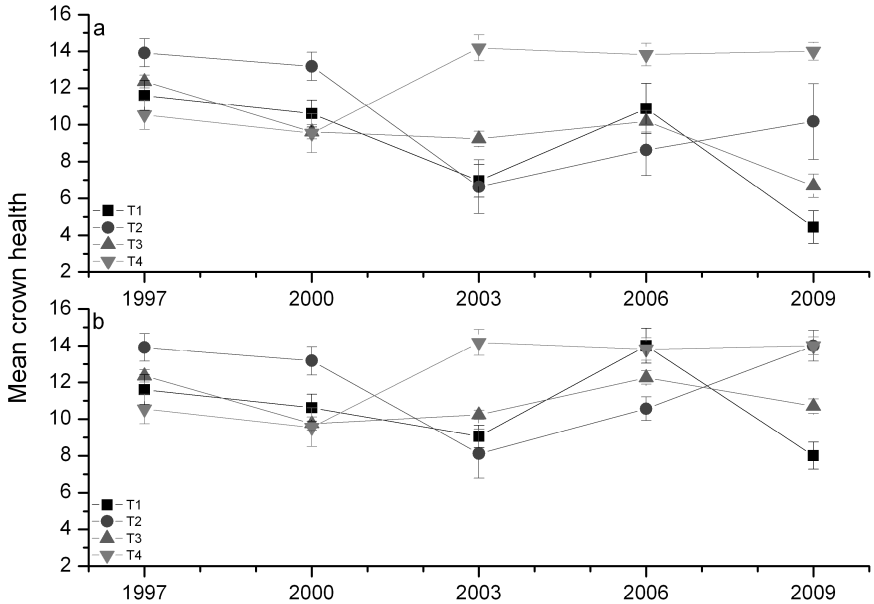

Figure 29. Melaleuca cuticularis *mean (+/- 1 SE) aggregate crown health scores (3 – extremely stressed, 23 - max health) for transects 1-4. (a) Means calculated by retaining trees that died after the first and subsequent sample years by assigning a zero condition score. (b) Means of trees extant at each sample time.*

Table 6. Chi-square values (N, df) for Friedman non-parametric ANOVA of tree health ratings over sampling times (1997, 2000, 2003, 2006 and 2009) and Wilcoxon paired test comparing 1997 to 2009. The first in each couplet of test results includes zero scores for dead trees. The second test statistic only includes individuals that were alive at all sample times .Where one test result is shown all trees survived.

	Transect 1	**Transect 2**	**Transect 3**	**Transect 4**
Freidman ANOVA χ-square (N, df)				
Melaleuca cuticularis	34.57 (25, 4) +++	19.56 (11, 4) ++	69.29 (67, 4) +++	18.38 (8, 4) ++
	15.70 (15, 4) ++	15.37 (8, 4) +	46.70 (44, 4) +++	
Melaleuca brevifolia				28.05 (60, 4) +++
				33.06 (53, 4) +++
Eucalyptus incrassata				69.23 (54,4) +++
				50.55 (36,4) +++
Wilcoxon paired test Z (N)				
Melaleuca cuticularis	3.77 (25) ++	2.58 (11) +	5.97 (74) +++	2.37 (9) +++
	2.38 (15) +	0.42 (8) ns	3.23 (48) ++	
Melaleuca brevifolia				1.88 (64) ns
				3.5 (59) +++
Eucalyptus incrassata				6.23 (54) +++
				5.04 (37) +++

P<0.05, +, P <0.005, ++, P <0.0005, +++,

DISCUSSION

Vegetation of the riparian zone.

The riparian vegetation of Lake Wheatfield has seen declines in the basal area and condition of *M. cuticularis* within the sub-littoral zone of the wetland with a strong relationship evident between elevation and decline. Similarly, *E. incrassata* has declined on low lying areas of the drainage channel. At elevations above the zone of prolonged inundation *M. cuticularis* has persisted. Other taxa occurring on elevated dunes have shown no decline although this is somewhat confounded by the occurrence of fires over the course of the study.

Over the study period wetland standing water depth fluctuated with rainfall, although there is evidence that prior to 1999 water levels were, on average, lower and it is hypothesised more seasonally variable with periods of drying of the current sub-littoral zone. The increase in groundwater level between 2000 and 2009 is suggestive of a coupling between wetland standing water depth and groundwater rise.

Degeneration of the mature population of *Melaleuca cuticularis* was evident at the south-east and east edge of the wetland where concentric rings of dead or dying trees formed a fringe around the basin edge where they occurred. This is likely due to a combination of existing salinity and recent prolonged water-logging creating anoxic conditions therefore drowning the trees.

Melaleuca can survive flooding with relatively fresh water (Froend *et. al.* 1987, Froend & van der Moezel 1994) to moderately salty water (EC of < 1000 mS m^{-1}). For example, *M. cuticularis* in the Peel-Harvey Estuary survive inundation of saline surface water to ~0.2 m above the soil surface (Carter 2006). Plants may not survive higher salinities (EC up to 6500 mS m^{-1}) where there is a combined stress of waterlogging, which can impair root function, and salinity (Barrett-Lennard 2003), and this is the probable cause of the degeneration of *M. cuticularis* stands in this study.

Melaleuca cuticularis has been observed to tolerate flooding at Coomalbidgup Swamp west of Esperance, although tolerance appeared to decline with prolonged flooding (up to 3 years) (Froend & van der Moezel 1994). In the glasshouse, Carter *et.al.* (2006) showed that when the external NaCl concentration increases, foliar organic solute concentrations in *M. cuticularis* leaves help counteract the salt (400 mM NaCl) for 22 days by keeping the ion concentration high in the organelles and therefore providing osmotic balance enabling the plant to 'exclude' these ions from its leaves. Analysis showed that after 18 days there was damage to the photosynthetic function of *M. cuticularis* in the 400 mM NaCl waterlogged treatment (Carter *et.al.* 2006).

These observations highlight the threats to the riparian vegetation at Lake Wheatfield. Accordingly, a management initiative aimed at lowering the standing water level of Lake Wheatfield and involving the construction of gravity pipeline between Lake Wheatfield and the ocean was implemented in October 2009 as component 1 of a wider plan (DEC 2005, Maunsell 2006, 2009b). Because the relationship between basal area change and elevation was positively correlated (*Melaleuca* declined over the study period at elevations less than ~1.45 m from the lowest point of the wetland) there is hope that the management intervention will result in improvement in the health of the riparian vegetation. Until 2009 the lake level was not recorded in spring below 1.5 m at the depth gauge whereas in November 2010 lake depth had fallen to 1.18 m. It is also important to consider the seasonal variability in water depth for management purposes rather than engineering for a lower depth that is relatively stable. Peak depths will be difficult to manage and engineering should legitimately focus on reducing water depths over summer to expose the sub-littoral zone and reduce anoxic conditions. It may take some years for the benefit of engineering interventions to become apparent.

Biodiversity components of the water body

Waterbirds

Long-term pattern
Waterbird species richness and guild structure were consistent throughout the monitoring period and indicative of a diverse community. The relatively stable conditions of water level, salinity and pH result in a similar fauna from year to year. Abundance was much more varied, but was not strongly correlated with any of the measured environmental parameters. Consequently, the current data offer little insight into factors driving waterbird usage at the wetland. Waterbirds move freely and extensively across the Warden wetlands (Halse 2007, Bennelongia 2008, Pinder 2010) and it is likely that abundance at Lake Wheatfield, particularly in autumn, is determined by the extent of water in other wetlands both within and outside of the Warden system. Individual feeding guilds were affected by water depth only modestly. Waders were most diverse at the lowest water levels when beaches were at their greatest extent, however they were never abundant. In contrast, diving species reliant on animal prey (e.g. the cormorants) were a diverse and abundant component of the fauna on most sampling occasions and occasions of low diversity (e.g. Autumn 1997 at 0.7 m and Late-winter 2005 at 2 m) do not correspond well with a particular depth regime.

Waterbird community structure has shown no consistent trend of change over the monitoring period. Rather, based on the biennial sampling, annual differences have been small and there is no suggestion that species observed in 1997 are unlikely to re-occur. Historical data (Jaensch *et al.* 1988) indicated lower annual species richness than the current program. It is likely, however, that the observed differences in richness are largely a function of different survey protocols. The historical surveys were conducted from the shore and would have had less access to all parts of the wetland compared to the surveys conducted by boat in the current study. Despite lower species richness the historical surveys included 5 species not recorded in the current program; Australasian Grebe (very rare in Warden wetlands generally), Banded Stilt, Clamorous Reed-warbler, Red-capped Plover and Fairy Tern. Only the latter two species were recorded in more than one of the 27 historical surveys suggesting most of these species were rare sightings. A Fairy Tern was observed on the nearby Windabout Lake in February 2010.

Three species recorded in all surveys of the current monitoring program were not observed at lake Wheatfield during surveys in the 1980s (Jaensch *et al.*1988). These were the White Ibis, Straw-necked Ibis and Yellow-billed Spoonbill. All three species are recent immigrants from northern parts of Western Australia, with 1952 marked as an important year for the spread and persistence of these species in the south-west (Serventy & Whittel 1967). With the exception of the Straw-necked Ibis these species were not recorded in the Esperance region during the Atlas of Australian Birds project (Blakkers *et al.* 1984). While the spread of these species in the south-west is in response to factors outside the Warden wetlands their successful breeding at Lake Wheatfield is likely to have been enhanced by the increasing water levels that have flooded beneath a large stand of *Melaleuca cuticularis* at the south-west side of the lake's main basin. Such flooding, while resulting in death of most of the trees, has created a well protected breeding site. This site is also utilised by Little Black Cormorant, Little Pied Cormorant and Darter. However, this breeding site may have a relatively short functional time span given the demise of the trees creating it.

Chestnut Teal
It has been estimated that in excess of one percent of the south west Australian population of Chestnut Teal may at times reside within the Warden Wetlands system (DEC 2009, Pinder et

al 2010; see also Halse 1994, 1995). Chestnut Teal were recorded at Lake Wheatfield in all waterbird surveys during the monitoring period.

The number of individual Chestnut Teal recorded during monitoring was greater than recorded in previous surveys by Jaensch *et al.* (1988). The highest abundance was 429 birds recorded in August 1997. Halse *et al.* (1994, 1995) present estimates of Chestnut Teal abundance in the Esperance region ranging from 918 (November 1989) to 31532 (March 1991). Clearly Lake Wheatfield supports only a small proportion of Chestnut Teal abundance at any one time. At Lake Wheatfield the abundance of this species appeared to be independent of season, while estimates of abundance for the Esperance region (Halse *et al.* 1992, 1994, 1995) indicated substantially higher numbers occurred in March than November. This congregation in autumn was observed for many species and is probably a response to declining water levels in many wetlands regionally. The absence of congregation at Lake Wheatfield suggests that use of the lake is opportunistic with the bulk of the local population occurring elsewhere.

Hooded Plover
The Hooded Plover was not recorded in any surveys during the study period nor has it been recorded at Lake Wheatfield in previous surveys (Jaensch *et al.* 1988, Pinder *et al.* 2010). Suitable habitat is not available at Lake Wheatfield where the shoreline is steep and short and would offer little opportunity to forage compared to the broad beaches that are preferred (e.g. Schulz et al 1984, Weston & Elgar 2000).

Optimum water depth
Increased water depth and the loss of shorelines and shallow feeding areas have been seen as a major threatening process to waterbird communities in the Warden wetlands generally. An analysis to determine the optimal depth of Warden wetlands in order to maximise species richness (Roberston and Massenbauer 2005) concluded that Lake Wheatfield waterbirds were less influenced by the loss of shallow habitats than at Lake Warden because the lake's bathymetry allows for only small areas of this habitat even at low water levels. The pattern of occurrence of waders during the monitoring period confirms that while a seasonal decline in water levels will increase annual species richness at Lake Wheatfield it will not provide substantial feeding areas for large numbers of shallow water specialists. The operation of the component 1 pipeline commenced in 2009 and resulted in similar autumn conditions of depth and salinity to those recorded in Autumn 1997. While there was an observed increase in the richness of small waders in February 2010, abundance was generally low and there is insufficient evidence to suggest a major change in community composition. It is likely that inter-related changes in sediment structure, invertebrate food sources and waterbird experience will take some years to re-establish after the lengthy period of sustained inundation. There is however, strong evidence that the pipeline has continued to seasonally lower water levels beyond this study period (John Lizamore –DEC Esperance-pers. com.). Consequently, future monitoring periods will be better able to measure the response of bird communities.

As discussed above, the condition of riparian vegetation has declined under sustained inundation. While this has assisted some species (e.g. Straw-necked Ibis and Yellow-billed Spoonbill) by providing nesting habitat these benefits are short-term and continued loss of riparian vegetation is likely to have a negative effect on shelter seeking species such as Grey and Chestnut Teal which nest and roost at the lake margin and would benefit from the persistence of closed canopy stands of *Melaleuca cuticularis* which are only seasonally flooded.

Invertebrates

High diversity

The diversity of invertebrates at Lake Wheatfield is high and indicative of generally healthy condition. Not only is the total species pool large but diversity at the family level is also significantwith more than 30 families on all sampling occasions. Jones *et al*. (2008) suggest a reference range of >10 families would indicate "best" condition for saline basin wetlands of the Avon region.

This diversity of species also reflects a uniqueness of the fauna as many recorded species are rarely encountered elsewhere in the wheatbelt. The assemblage I species defined by Pinder *et al.* (2004) may have limited distributions in Western Australia since there collection appears to be associated to wetlands and streams of the south coast. This assemblage at Lake Wheatfield includes *Melita kauerti* (Amphipoda) and *Ascorhis occidua* (Gastropoda), which may be more common in estuarine habitats, and *Symphitoneuria wheeleri* (Trichoptera). The colonial hydrozoans *Cordylophora sp*., collected in all but one sampling year, has not been collected at any other site in the DEC monitoring or survey program; which includes more than 370 wetlands across the south-west, Carnarvon Basin and Pilbara.

Water chemistry parameters were not correlated with species richness. However, the lowest species richness (42 species) in 2009 coincided with the lowest depth, highest EC and highest pH recorded in spring; when invertebrate samples were collected. Additionally, a bloom of diatoms (possibly *Chaetocerus* sp.) was apparent at sub-site A, causing the water at this site to have a syrupy quality. The combination of these factors is unlikely to encourage invertebrate colonisation and would limit species richness. By contrast, the highest richness (1999) occurred at the highest depth, when EC was at its lowest and pH was close to the median; conditions highly favourable for optimum development of invertebrate communities.

Individually, the majority of water chemistry parameters had only small apparent effects or no effect on invertebrate community composition. The measured ranges of salinity (EC), pH, and alkalinity, at the time of invertebrate sampling, were small and likely to be well within the tolerances of the majority of invertebrates collected. Concentrations of phosphorus and chlorophyll were heavily skewed by a few large observations, but this may have influenced community composition. It is likely that the cumulative history of these parameters leading up to the sampling period was the greatest determinant of species composition. Communities collected at low salinity and pH also had lower soluble phosphorus concentrations and a tendency to include more insects and macroinvertebrates, while those with constant, albeit slightly higher, salinity had higher soluble phosphorus and tended to include more microinvertebrates, particularly rotifers. This shift may have accompanied a change in functional dominance between the benthos and water column; with greater productivity in the water column when microinvertebrates are dominant and responding to phytoplankton driven by high phosphorus concentrations, versus greater benthic production when macroinvertebrates are dominant. Cook and Farrell (2008) recorded lower invertebrate biomass in Lake Wheatfield in 2007 compared to 2006 (Cook *et al.* 2006) and this coincided with a shift away from macroinvertebrate dominance of standing biomass. The frequency or period over which dominance might change, cannot be determined from the biennial sampling protocol of the monitoring program, however, it is clear that dominance can change backwards and forwards and does not imply a long term directional change in the invertebrate community.

Black Bream (*Acanthopagrus butcheri*) is a common species within Lake Wheatfield. While there are no data to identify how this species may affect invertebrate communities at Lake Wheatfield, it is known to prey on the predominant invertebrate present and to show a marked change in diet with increased size (Sarre 1999). Both of these factors would suggest the potential to actively structure invertebrate communities. This fish is also likely to be an important food resource for the many fish eating waterbirds observed at Lake Wheatfield. Information on population dynamics for this fish may be important in correctly interpreting both invertebrate and waterbird data.

Temporal species turnover vs. change
The estimated species pool (i.e. the entire suite of species available to establish populations within the wetland) is in excess of 200 species based on the sampling to date. This clearly indicates that the wetland supports many more species over time than are encountered on any single occasion and 30% more than have been present in all our samples to date.

The concern is that this apparent diversity may be the result of a directional species turnover in response to environmental change. The large and constant number of singleton species observed would be consistent with sampling from a community undergoing directional change, with new species added in successive years. However, this would imply a concomitant loss of species which had already been recorded. Such was not the case. With the exception of singletons (the richness of which remained constant), there are only 5 species which were collected in the period 1997-2001 and not also collected in the period 2003-2009.

There are several alternative hypotheses to explain the high proportion of singleton species (e.g. many species of low abundance are rarely collected; a large number of species in the regional species pool only rarely occur in the lake; a sampling artefact). While none can be discounted it seems likely that the high incidence of singletons is the result of sampling a community with large intra-annual species turnover reflecting a successional or stochastic change in community structure throughout the year. With sampling tightly constrained by the calendar, but community development depending on annual conditions (timing of rainfall, temperature, etc) different stages within the development of the community may be sampled in different years. This should be a reminder that substantially different communities could occur outside the sampling period thus greatly enhancing the overall invertebrate diversity of the wetland. Active management to lower water levels (e.g. the stage 1 pipeline) is unlikely to impact on these invertebrate communities provided the annual hydroperiod includes a range of water depths and salinities, similar to those already observed, in which these diverse communities can periodically develop.

Concluding remarks

The various components and processes measured and discussed for Lake Wheatfield are summarized in a conceptual model (Fig. 31) which is essentially a collection of hypotheses about linkages between wetland components. These hypotheses are consistent with the collected data, but undoubtedly reflect only a subset of the processes affecting the wetland. Central to our understanding of how components are linked is the view that during the monitoring period the wetland was essentially stable, with a hydroperiod that varied between years, but not sufficiently to result in major changes in wetland conditions. That we have not always sampled at the same stage of the hydrological regime has probably exaggerated differences between years, but essentially the system was similar from year to year. This was reflected in the richness and community composition of waterbirds. Waterbird abundance was more variable and poorly correlated with the parameters measured during monitoring. It is probable that abundance is dependent on factors outside the wetland such as the prevalence of water in other Warden wetlands and their suitability for feeding.

While invertebrate and waterbird communities have remained fairly stable the riparian vegetation of Lake Wheatfield has shown significant decline over the course of the study. Management actions aimed at increasing the magnitude of seasonal drying to improve vegetation condition are not in conflict with management of the lake for the faunal communities of the waterbody.

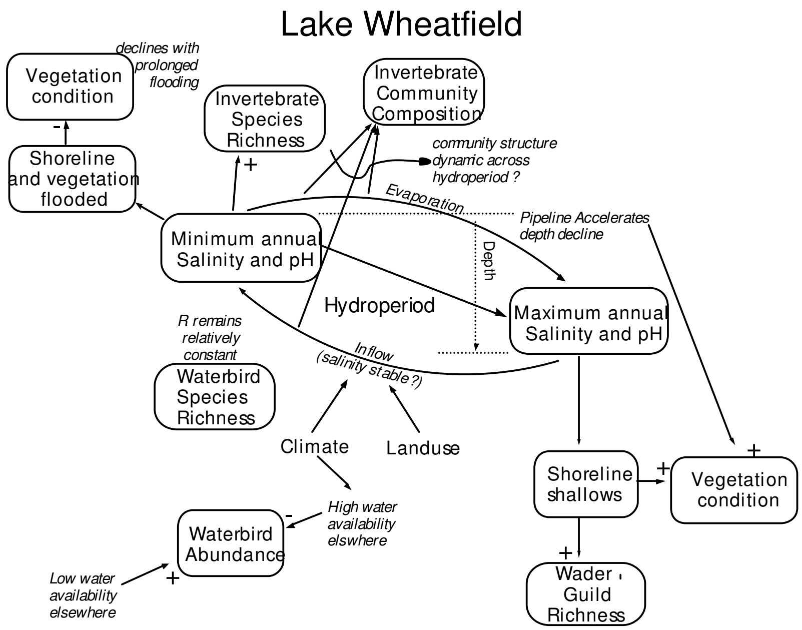

Figure 30. Conceptual model of lake biodiversity components and their interactions

ACKNOWLEDGMENTS

Funding for this monitoring program is provided by the State Salinity Strategy via the Natural Resources Management Branch of DEC. Salinity, pH and wetland depth gauge data were provided by Jim Lane, Yvonne Winchcombe and Alan Clarke of DEC. Observation bore (groundwater) data; depth, pH and conductivity were provided by John Lizamore and Tilo Massenbauer, also from DEC. Edyta Jasinska and Melita Pennifold processed and identified invertebrate samples. Stuart Halse identified Ostracoda and Russ Shiel identified Cladocera and Rotifera.

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APPENDICES

Appendix A.1 *Water chemistry parameters for Site A at Lake Wheatfield 1997-1999 monitoring years.*						
Date	5/08/97	17/11/97	10/03/98	26/08/99	19/10/99	23/03/00
Monitoring Year	1997	1997	1997	1999	1999	1999
Depth (m)		2	0.7	2.5	2.5	2
EC (µS/cm)	14870	10900	24000	8730	9010	5050
pH	8.15	8.53		8.19	8.69	7.52
TFN(µg/L)		2200		1600	1600	2300
TFP(µg/L)		20		30	20	50
Cphyll-a (µg/L)		46		32	22	13
Cphyll-b (µg/L)		6		2	5	3
Cphyll-c (µg/L)		4		2	2	5
Pphytin-a (µg/L)		21		5	<1	11
Temp (℃)		22		14.5	20.8	19.7
Diss.Oxy.(%)	93.4	159		104	121.8	41
Turbidity (NTU)		5.5			3.5	
Colour (TCU)		130			140	
TDS (g/L)		6.7			5.5	
Alkalinity (mg/L)		220			140	
Hardness (mg/L)		1100			880	
Si (mg/L)		11			11	
Na (mg/L)		2100			1700	
Ca (mg/L)		65			52	
Mg (mg/L)		220			181	
K (mg/L)		54			42	
Mn (mg/L)		0.01			0.01	
Cl (mg/L)		3500			2900	
HCO_3 (mg/L)		270			171	
CO_3 (mg/L)		1			1	
NO_3 (mg/L)		0.21		0.02	0.01	0.07
SO_4 (mg/L)		480			396	

Appendix A.2 *Water chemistry parameters for Site A at Lake Wheatfield 2001-2009 monitoring years.*

Date	21/08/01	23/10/01	27/03/02	13/08/03	22/10/03	29/03/04	9/08/05	26/10/05	24/03/06	8/08/07	23/10/07	1/04/08	26/08/09	27/10/09	25/03/10
Monitoring Year	2001	2001	2001	2003	2003	2003	2005	2005	2005	2007	2007	2007	2009	2009	2009
Depth (m)	1.77	1.92	1.34	1.78	1.91	1.36	1.65	2.01	1.43	1.81	1.87	1.2	1.7	1.58	0.82
EC (μS/cm)	11850	9080	15590	11200	9340	14070	8560	9260	12580	11580	11250	16990	12780	12610	24400
pH	8.45	8.32	8.63	6.69	8.02	8.33	7.28	7.76	8.69		9.1	9.16	8.3	9.35	8.88
TFN(μg/L)	1200	1400	1900	1400	1400	2000	1200	2100	3700	1400	4600	2400	2500	1300	
TFP(μg/L)	10	20	30	<10	<10	<10	20	30	30	10	700	30	480	<10	
Cphyll-a (μg/L)	21	13	21	16	11	27	4	2	0.5	4	8	9	2	<1	
Cphyll-b (μg/L)	<1	<1	2	2	2	<1	1	<1	19	<1	2	2	<1	<1	
Cphyll-c (μg/L)	<1	1	2	2	4	4	1	<1	5	2	2	4	<1	<1	
Pphytin-a (μg/L)	8	8	18	6	6	1	6	1	52	5	14	9	<1	<1	
Temp (°C)	17.2	17	20	13.8	19.6	18	13.7	17.8	19.8		16.2	19.6	13.7	20.4	27.8
Diss.Oxy.(%)	124						85.5	96.2	102		89.3				
Turbidity (NTU)		15			0.7			2.4			1.1			30	
Colour (TCU)		96			130			160			32			48	
TDS (g/L)		5.4			5.3			6.4			6.6			7.9	
Alkalinity (mg/L)		130			128			125			150			160	
Hardness (mg/L)		820			900			980			1100			1200	
Si (mg/L)		12			6			8			1.4			0.3	
Na (mg/L)		1570			1720			1910			2180			2530	
Ca (mg/L)		47			55			53.7			58.3			62.6	
Mg (mg/L)		170			186			206			231			253	
K (mg/L)		43			40.7			50.7			51.3			72	
Mn (mg/L)		0.01			0.0005			0.003			0.0005			0.0005	
Cl (mg/L)		2500			2800			3170			3420			4190	
HCO_3 (mg/L)		159			156			153			73			195	
CO_3 (mg/L)		1			1			1			54			<1	

Appendix A.2 Water chemistry parameters for Site A at Lake Wheatfield 2001-2009 monitoring years.															
Date	21/08/01	23/10/01	27/03/02	13/08/03	22/10/03	29/03/04	9/08/05	26/10/05	24/03/06	8/08/07	23/10/07	1/04/08	26/08/09	27/10/09	25/03/10
Monitoring Year	2001	2001	2001	2003	2003	2003	2005	2005	2005	2007	2007	2007	2009	2009	2009
NO_3 (mg/L)	0.01	0.01	<0.01		0.01		0.04	0.11		<0.01	0.01	0.01	0.03	0.01	
SO_4 (mg/L)		407			408			448			534			594	

Appendix A.3 *Overstorey species list for trees monitored at Lake Wheatfield*			
T1 (management burn, 2009)	**T2** (wildfire, 2003)	**T3** (inundated)	**T4** (peninsula)
Acacia cyclops			*Acacia sp.2*
Acacia saligna	*Acacia saligna*		*Eucalyptus incrassata*
	Banksia speciosa		*Eucalyptus occidentalis*
Melaleuca cuticularis	***Melaleuca cuticularis***	***Melaleuca cuticularis***	***Melaleuca cuticularis***
Melaleuca brevifolia			*Melaleuca brevifolia*
Nuytsia floribunda			
Spyridium globulosum			
Previously but no longer encountered species were *Nuytsia floribunda at* T1 and *Acacia* sp.2 at T4.			