Waterbird and invertebrate communities at Toolibin and Walbyring lakes during the 2017 fill event

David Cale and Adrian Pinder

April 2019

GOVERNMENT OF WESTERN AUSTRALIA
Department of **Biodiversity, Conservation and Attractions**

Biodiversity and Conservation Science

Department of Biodiversity, Conservation and Attractions
Locked Bag 104
Bentley Delivery Centre WA 6983
Phone: (08) 9219 9000
Fax: (08) 9334 0498

www.dbca.wa.gov.au

April 2019

Questions regarding the use of this material should be directed to:
Principal Research Scientist
Ecosystems Science Program
Department of Biodiversity, Conservation and Attractions
Locked Bag 104
Bentley Delivery Centre WA 6983
Phone: 9219 9000

The recommended reference for this publication is:
Cale DJ and Pinder AM. 2019. Waterbird and invertebrate communities at Toolibin and Walbyring lakes during the 2017 fill event, Department of Biodiversity, Conservation and Attractions, Perth.

Cover image:
Flooded swamp sheoak at the southern end of Toolibin Lake in October 2017, (photographed by D Cale)

Table of Contents

Key points:

- Aquatic invertebrate richness in Toolibin Lake was very low in April 2017 and even though it had risen by October (to be about the same as in December 1996) it remained much lower than expected considering the very low salinity. Possible reasons for this are low dissolved oxygen concentrations, low viability of drought tolerant crustacean egg banks and few sources of freshwater colonists in the surrounding landscape.
- Waterbird richness in 2017 was within the range recorded by numerous surveys in the 1980s and comparable to data from 1996. There was no evidence of compositional change since 1996.
- Most of the six target waterbird species were present and three were breeding. The abundance of most target species was below the management target range. However, given similar low abundance in some surveys from the 1980s, it is unclear whether a failure to meet abundance targets on any single occasion reflects a diminished capacity of the wetland or simply periodically suboptimal conditions.

Summary

In February 2017, rainfall in excess of 100mm across its catchment resulted in the largest fill event for Toolibin Lake since active management of the wetland's hydrology began in 1994. The lowering of water tables below the lake bed and the diversion, past the lake, of inflow water in excess of ~1000mg/l total dissolved solids (TDS) has resulted in regeneration of vegetation on the lake bed. However, in conjunction with reduced rainfall the management of surface water contributed to fewer, shallower and shorter inundations in the lake such that the lake was inundated <15% of the time between 1996 and 2016 compared to an estimated 70% of the time historically.

This study presents an analysis of waterbird and invertebrate community structure during the 2017 inundation of Toolibin Lake, with the aim of 1) comparing it with a) available information (largely anecdotal) about community structure before management of the lake's hydrology began in 1994, and b) with a more recent detailed survey conducted in 1996 before the relatively drier period over the last two decades and 2) to compare community structure in 2017 against management goals.

Inundation of the lake in 2017 resulted in a peak depth of 2.24m; 1m greater than occurred in 1996. Water quality and the physico-chemical environment (particularly of invertebrates) were very different in 2017 compared to 1996. Salinity (TDS) was considerably lower, increasing from 630 mg/l in April to 780mg/l in October 2017, compared to 9400 mg/l in December 1996. Dissolved oxygen was at low concentration (<36% saturation) compared to the 200% saturation reported in 1996. There is evidence that primary production was primarily benthic in 1996 and primarily in the water column in 2017. Many of the differences in water chemistry in 2017 may be attributed to the breakdown of large volumes of 'leaf litter' accumulated in the years prior to the inundation.

Toolibin Lake supported 35 and 57 species of macroinvertebrate in April and October 2017 respectively. This is higher than the richness observed at Walbyring Lake (22 and 33 species recorded for the same periods) and is a reversal of the previously described relative invertebrate richness of the two wetlands. It is likely that the significantly lower salinity at

Toolibin Lake in 2017, compared to 1996, contributed to a relative increase in diversity in this lake and to a greater similarity of composition between the two lakes.

Invertebrate diversity in Toolibin Lake was very low in April 2017, most likely due to a combination of water quality (particularly low dissolved oxygen) and a lag time for colonisation. There is some evidence that even in October the development of the community was retarded, with richness lower (by about 20 species on average) compared to other Wheatbelt wetlands of similar salinity. Several features of the invertebrate community, including the low richness of ostracods and freshwater species and the high proportion of highly dispersive insects, suggest that sources of colonists were compromised. Egg banks of resting stages of invertebrates such as ostracods and cladocerans within the wetlands' sediments may have been depleted since the late 1990s if the minor inundation events over the intervening 20 years were not sufficient to replenish them. It could also be the case that the condition of neighbouring wetlands has declined and now supports fewer potential colonising species.

From a management perspective, at salinities < 1000mg/l, the conservation value of current invertebrate communities in Toolibin Lake and Walbyring Lake are similar. However, these communities comprise widely distributed species and are likely to have greater importance in support of the waterbird community rather than any inherent conservation value.

During 2017 a total of 19 species of waterbirds were recorded at Toolibin Lake and 16 at Walbyring. The richness of individual surveys in 2017 was within the range recorded by numerous surveys in the 1980s and comparable to data from 1996.

There has been some compositional change in waterbird communities since the 1970s. Australasian bittern and purple swamphen were affected by the loss of reed habitats and were no longer using the lake in the 1980s. Changes in community composition since the 1980s include reduced diversity of the diving (cormorants and grebes) and large wader (herons, egrets and spoonbills) guilds. The reduced number of species in these guilds was already established in 1996 and persisted in 2017 and it is likely that the loss, since at least the 1990s, of large prey species including fish, tadpoles and freshwater crayfish and shrimp was critical. There is no evidence that differences in the invertebrate communities between 1996 and 2017 further restricted the development of waterbird communities.

In 2017, key species such as the freckled duck continue to be present at Toolibin Lake and five species were recorded breeding (with an additional species at Walbyring Lake) which is comparable to the 1-6 species recorded in four pre-salinity baseline surveys from the 1980s. All but one of the six waterbird indicator species listed in the Toolibin Lake recovery plan were present and three of them were recorded breeding. The absence of black-winged stilt, the sixth indicator species, is not of concern because all surveys were conducted when the lake was considerably deeper than preferred by this species. The abundance of all but one indicator species (grey teal) was below the desired management target range. However, given similar low abundance in some surveys from the 1980s, it is unclear whether a failure to meet abundance targets on any single occasion reflects a diminished capacity of the wetland or simply periodically suboptimal conditions.

Introduction

Toolibin Lake lies 180 km south east of Perth within the Wheatbelt of the south west of Western Australia. Together with the superficially similar Walbyring Lake, immediately downstream, Toolibin lies within a 1230 ha system of Class “A” nature reserves (Toolibin Lake Recovery Team and Toolibin Lake Technical Advisory Group, 1994) at the headwaters of the Northern Arthur River (DBCA, 2017). The wetlands in these reserves are actively managed by the Western Australian Department of Biodiversity, Conservation and Attractions for their significant biodiversity value.

The lake is an important example of the threatened ecological community (TEC) comprising “perched wetlands, in the Wheatbelt region, with extensive stands of living swamp sheoak (*Casuarina obesa*) and paperbark (*Melaleuca strobophylla*) across the lake bed” (e.g. Hamilton-Brown & Blyth, 2000). This wetland type is characterised by a flat lake bed with an overstorey of *Melaleuca* and taller sheoak and a dependence on seasonal, temporary inundation with freshwater. While once widespread, salinisation as a result of land clearing has degraded at least 90% of the area and examples of this TEC (*op.cit*).

The TEC is particularly important habitat for waterbirds. At Toolibin Lake, data for the occurrence of waterbird species have been collected since the 1970s (see Froend & Storey, 1996 for a review), with 50 species recorded in total (McMahon, 2006). A series of 49 surveys of waterbirds conducted between 1981 and 1985, as part of the “Waterbirds in Nature Reserves Study” (Jaensch, Vervest & Hewish, 1988), recorded 41 species, ranking Toolibin Lake as 13th when compared with 197 other wetlands surveyed during the same period. Twenty two species were recorded breeding by these authors, ranking 1st amongst the 197 studied wetlands and representing the highest published richness of breeding species in wetlands of the south-west region of Western Australia. Lake Walbyring was included in these surveys but supported a less diverse assemblage of species; total richness was 19 species with 7 species breeding, giving ranks of 55 and 23 respectively (Jaensch *et al.*, 1988). Halse (1987) attributed the importance of Toolibin Lake as a breeding site to three factors: 1) the stands of inundated paperbark and sheoak, 2) water quality sufficiently fresh to support this vegetation and very young waterbirds and 3) periodic drying which again supported the vegetation and also increased the production of food for breeding birds.

Breeding of several waterbird species at Toolibin Lake during the 1980s was particularly significant because there were few breeding sites for these species throughout the south west. These included great cormorant for which Toolibin Lake was the only recorded breeding site, freckled duck which bred in only 4 wetlands and little pied cormorant, little black cormorant, blue-billed duck, white-necked heron, great egret, rufous night heron and yellow-billed spoonbill, all of which bred in <10 of the 197 studied wetlands (Jaensch *et al.*, 1988).

The conservation value of the TEC, the waterbird assemblages and the importance as a breeding site were integral to the listing of Toolibin Lake as a wetland of international importance under the Ramsar Convention.

Aquatic invertebrate communities are integral to the successful breeding of waterbird species providing food for females preparatory to breeding and meeting protein requirements of juveniles during their rapid early growth (Frith, 1959; Maher & Carpenter, 1984). Two previous studies have documented the community structure of aquatic invertebrates at both Toolibin Lake and Walbyring Lake. Doupé and Horwitz (1995 Appendix 1) collected 36 species at Toolibin Lake in September 1992 and 31 species at Walbyring Lake. The collected fauna was dominated by aquatic insects and a large proportion of the richness at each lake was not collected in the other. In December 1996 Halse *et al.* (2000) collected 52 and 63 species at Toolibin Lake and Walbyring Lake respectively and suggested that the higher richness (a product of greater sampling effort) was a better reflection of the conservation value of the wetlands.

Land clearing for agriculture has caused salinization of many Wheatbelt wetlands and was recognised as a primary threat to the persistence of Toolibin Lake and the associated fauna and flora (Northern Arthur River Wetlands Committee, 1987). Toolibin Lake is the last major natural wetland, within a chain of wetlands in the headwaters of the Arthur River, that has not become saline (McMahon, 2006). Most of the wetlands in this chain are significantly degraded by secondary salinity, from land clearing for agriculture, which has resulted in increased salt loads from surface inflows and direct interaction with groundwater beneath the lake bed. Salinisation had already affected some overstorey vegetation elements at Toolibin Lake by the 1980s when capillary action from rising groundwater increased soil salinity when the lake was dry and caused a loss of vigour amongst even the most tolerant species (Froend *et al.*, 1987). The loss of some fringing beds of sedges may have occurred during the 1970s (Casson, 1988; Froend, Halse & Storey, 1997). Before land clearing in the catchment Toolibin Lake was an ephemeral fresh to brackish wetland and believed to fill in seven out of ten years (Stokes & Sheridan, 1985; Rutherford, 2015). The lake typically filled with seasonal rain and then retained water, even in the absence of further inflow, into a second year, albeit with a decline in depth and breeding of waterbirds (Casson, 1988).

Engineering solutions for the management of saline water in the Lake Toolibin catchment were implemented from 1994 (DBCA, 2017) and include:

- Groundwater pumping stations to lower the water table below the lake and reduce the interaction of saline groundwater with the lake bed and surface water.
- A diversion bank which can allow entry of high volume, fresh surface inflows (<2000 μS/cm) to the lake, but divert saline flows past Toolibin and Walbyring Lakes and into saline wetlands further downstream.
- A system of sump, channels and pump on the lake bed which can be used to remove surface water in the drying phase of the lake, before evapo-concentration increases salinity above 10 mS/cm, thus preventing the accumulation of salt and vegetation death.

There is evidence that these solutions have had a positive effect; with waterbird usage continuing when the wetland fills, regeneration of lake bed vegetation and an increase in the depth to groundwater (DBCA, 2017).

However, maintenance of water quality has come at some cost. The diversion bank has successfully diverted all but the freshest inflows since 1994 and in conjunction with declining rainfall (e.g. Froend *et al.*, 1997; Muirden & Coleman, 2014) this has meant a reduced volume and frequency of inflow (e.g. Muirden & Coleman, 2014; Table 4.9). While there have been numerous small fill events since 1994 (e.g. 2008 and 2012) these have inundated only portions of the lake bed and persisted for only a few months. Sufficient inflow to raise lake levels and maintain them for a period approaching 12 months occurred in 1996 and 2006 when the wetland was filled to a depth of ca 1 m (Bowra & Wallace, 1996) and 0.96 m (Bourke & Rutherford, 2018) respectively. In each of these fill events the wetland dried within 12 months of filling compared to hydroperiods typically approaching 2 years for earlier fill events (Casson, 1988 pg. 5; Lane, Clarke & Winchcombe, 2017). Consequently, rather than the wetland being inundated 70% of the time (e.g. Froend *et al.*, 1997) inundation occupied < 15% of the 20 year period from 1996 to 2016, including two decade-long periods without significant inundation.

In February 2017, Toolibin Lake filled to a peak depth of 2.24 m, with a volume of 4000 ML and an inundated area of approximately 273 Ha (Bourke & Rutherford, 2018). This fill event was the most substantial since the 1980s presenting the first opportunity to compare waterbird and invertebrate community structure at levels of inundation only encountered prior to active management of the wetland's hydrology. This study presents an analysis of waterbird and invertebrate community structure during the 2017 inundation of Toolibin Lake, with the aim of 1) comparing it with available information about community structure before management of the lake's hydrology began in 1994 and in 1996 (i.e. Halse *et al.*, 2000) and before the relatively drier period over the last two decades and 2) to compare community structure in 2017 against management goals.

Methods

Toolibin and Walbyring Lakes commenced filling on February 9 2017, as the result of rainfall in excess of 100 mm over the 72 hour period to February 12 (Bourke & Rutherford, 2018). The water level in Toolibin Lake peaked at 2.24 m; the highest water level since construction of the diversion channel and 1 m greater than the previous highest level, of 1.1 m in July 1996 (Bourke & Rutherford, 2018).

Both wetlands were visited on April 11-12 (approximately 8 weeks post filling) and October 27-28 (approximately 36 weeks post filling) to assess the community structure of waterbirds and aquatic invertebrates.

Three sites were sampled for invertebrates in each of Toolibin and Walbyring lakes. These sampling sites approximately matched the location of sites used by Halse *et al*. (2000). Site Toolibin West (W) was on the western side of the lake (32.91944082° S 117.5992981° E) south of the small inflow creek on that side and coincidentally near a point at which the separator bank was breached during the 2017 fill event. This site had higher salinity in both 1992 (Doupé & Horwitz, 1995) and 1996 (Halse *et al*., 2000) compared to a site on the eastern shore. The area sampled was deeper than other sites with less standing and fallen timber and only scattered live trees. The site Toolibin East (E) was situated on the eastern shore (32.92149 ° S 117.6166 ° E) in the vicinity of the depth gauge maintained by the South West Wetlands Monitoring Program (SWMMP) since 1978 (Lane *et al*., 2017). This site had many submerged logs and included scattered standing live and dead trees but did not extend into dense stands of living trees. Toolibin South (S) on the southern shore (32.92983572° S 117.6095301° E) included areas of dense live trees and more open areas of scattered trees but few submerged logs. It was of a similar depth to the eastern site. In April 2017, the first site in Walbyring Lake (W1a) was located on the south shore (32.93765 ° S 117.5896 ° E) approximately 400 m west of site W2, and was physically similar to W2 (see below). However, the site was relocated (W1b) to the eastern shore (32.93628023° S 117.5961924° E) in October 2017, thus sampling a different sector of the lake and more accurately reflecting the location of the sampling site of Halse et al. (2000). Site W1a and W1b had similar depth and composition of logs and standing trees but W1b was further from the road and closer to the drainage line connecting to Toolibin Lake. In October, W1b had a 2-3 m wide “raft” of *Lemna* sp. along the shore. Site Walbyring 2 (W2) was located in the vicinity of the SWMMP maintained gauge on the southern shore of Walbyring Lake (32.93902882° S 117.5939556° E). This area was deep relative to other sites in the lake and included scattered live and dead trees. In October 2017 the shore line included a 0.5 m wide “raft” of *Lemna* sp. Walbyring 3 (W3) was situated on the north shore (32.9343627° S 117.5877944° E) and was shallower than other sites. The site included many submerged or partially submerged logs but few standing trees. Unlike the other two sites there was no raft of *Lemna* sp. observed during October.

Waterbird surveys

Waterbirds were surveyed on 11/4/2017 and 28/10/2017 at Toolibin Lake and on 12/4/2017 and 29/10/2017 at Walbyring Lake. All surveys were conducted by boat, with two observers and a ‘skipper’ in April and a single observer in October. Care was taken to investigate all areas of the lakes and observers were confident that all species present were recorded;

however, abundance within dense stands of live trees in Toolibin Lake was likely underestimated. All broods observed were recorded as evidence of breeding, however no active searching for nests or other signs of breeding was attempted.

Long term trends in community structure were investigated using a dataset constructed from surveys from the 1970s, 1980s and 1996 in conjunction with the 2017 surveys. Waterbird surveys from the 1980s (1981-1984 and 1987) were extracted for Toolibin Lake (45 surveys) and Walbyring Lake (16) from an unpublished database generated by the Waterbirds in Nature Reserves study (Jaensch *et al.*, 1988). All surveys that included waterbirds were included in the dataset. At Toolibin Lake these surveys included 4 from spring and 1 from autumn which were used as a "pre-salinity baseline" by Froend et al. (1997 Table 1). The remainder spanned 11 of the 12 calendar months between 1981 and 1984, including 11 from April, October and December (op. cit.). To represent the community in the 1990s, a survey for each lake conducted on December 18 (Halse *et al.*, 2000) was incorporated into the dataset. Finally 2 data sets from the 1970s were included. The first, a species list from numerous surveys conducted during the period 1965-1975 (Goodsell, Garstone & Lambert, 1978) was presence/absence only and, because it is cumulative, represents a very different sampling effort from individual surveys. The second 1970s survey was again presence/absence only, but represents the birds present on 31/10/1975 (Munro, 1975).

Depth and salinity for 1980s bird surveys were available for dates within 10 days of 21 of the analysed surveys (unpublished data from SWWMP Jim Lane) and data for the 1996 survey has been published (Halse *et al.*, 2000). These data have been used to assist interpretation of waterbird data where required.

Invertebrate sampling

Invertebrates were collected as two sub samples at each sampling site. Firstly, a sample was collected using a standard FBA D-framed net with a 53 µm mesh to gently sweep the water column and submerged surfaces over a disjunct 50m sampling path that included all recognisable microhabitats within an arc encompassing approximately 200 m of shoreline. A second sample was collected with a 250 µm mesh using vigorous sweeping of surfaces and 'heal kicks' of benthic substrates to collect larger, faster species and ensure collection of sediment dwelling species. These paired samples were preserved in the field in 100% alcohol and processed in the laboratory. Processing involved 1) sieving samples with a graded series of sieves to assist in the detection of all species under a binocular microscope at 10-40x magnification and 2) the removal of sufficient representative specimens of each taxon to ensure identification of all species. Specimens from the two subsamples were combined to yield a single sample of community structure for each site. The efficacy of this sampling protocol has been described elsewhere (Halse *et al.*, 2002; Pinder *et al.*, 2010). The sampling protocol only differs from the methods used at Toolibin Lake by Halse et al. (2000) in the use of a 53 µm rather than 110 µm mesh size. However, the sampling protocol represents a greater sampling effort than reported by Doupé and Horwitz (1995) who used a single sample of 500 µm mesh size. For 2017 and 1996 invertebrate data the abundance of each Invertebrate species is qualitative and based on an estimate of log class abundance i.e. 1-10 individuals is log-class 1, 11-100 individuals as log-class 2 etc. For the 1992 data (i.e. Doupé & Horwitz, 1995) only presence absence was available.

To investigate the diversity of micro-invertebrate assemblages in the two lakes the 53µm net samples collected in April were passed through a 250µm sieve and the <250µm portion was processed by Dr R Shiel (University of Adelaide) by subsampling to collect the first 200 individuals which were then identified and counted to give an approximate percentage relative abundance.

Water chemistry

To assist in the interpretation of invertebrate and waterbird community structure, water samples were collected to determine water quality parameters at each invertebrate sampling site, on both sampling occasions. A sample was filtered through a 0.45 µm filter for later determination of total filtered soluble nitrogen (TFN) and phosphate (TFP) and a measured volume of water was filtered through a standard microfibre filter for measurement of photosynthetic pigments. Additionally, water samples were collected from W2 at Walbyring Lake and W (April 2017) or E (October 2017) at Toolibin Lake to measure ionic composition, alkalinity, total dissolved solids (TDS), colour, turbidity and nitrate. All water samples were processed by The Chemistry Centre of WA using standard APHA methods; ionic composition (method; iMET1WCICP), Total Dissolved Solids (iSOL1WDGR), Alkalinity, Carbonate ion, Bicarbonate ion (iALK1WATI), Chlorophyll a, b c and phaeophytin (eCHLA1WACO), Colour,(iCOL1WACO), Nitrate (iNTAN1WCALC), TFN (iNP1WDFIA), TFP (iPP1WDFIA) and Turbidity (iTURB1WCZZ).

Field measurements of electrical conductivity (ec), pH and temperature (T) were made in situ at each sampling site with a TPS WP-81 portable meter.

Analyses

Community structure in both waterbird and invertebrate communities was analysed in terms of two metrics; species richness and community composition (including abundance).

Richness of the entire waterbird community was calculated for each survey in the dataset and used to calculate the median and inter quartile range of richness within the 1980, 1990 and 2017 time periods (method Boxplot; R Core Team, 2014). Because of the differences in sampling effort during the 1970s surveys, individual richness values are reported. The richness within waterbird guilds (defined in Table 4 of Halse, 1987) was investigated at Toolibin Lake in the same way as total richness. However, only 1980s surveys of 10 or more species (75% of surveys) were included with surveys for 1975, 1996 and 2017 when determining guild richness. This ensured a relatively complete guild structure was possible for the 1980s and made comparison across time periods more reasonable. Richness of invertebrate assemblages were dealt with in the same way as for waterbirds, with median and interquartile ranges calculated from the replicate sampling sites in each lake.

The abundance of 6 indicator waterbird species (DBCA, 2017 pg 27) was qualitatively compared across time periods by calculating the median and inter quartile range of raw abundance for each species within the 1980, 1990 and 2017 time periods (method Boxplot; R Core Team, 2014). All surveys from the 1980s were included in the analysis but no abundance data were available for the 1970s surveys. Abundance data were not standardised because this would not enable comparison with management goals described in terms of raw abundance (DBCA, 2017 Table 3).

Waterbird community composition was compared across surveys (Legendre & Gallagher, 2001) using methods in the *vegan* package (Oksanen *et al.*, 2013). An un-weighted paired group method (UPGMA) classification of a Bray-Curtis dissimilarity matrix for the community composition data was used to define objective groups of surveys based on similar composition. Analysis of similarity (ANOSIM; Oksanen *et al.*, 2013) was used to determine an appropriate number of groups for classifications and indicated that: above 8 groups dispersion within the main groups was little affected and new groups were comprised of single surveys, while fewer than 8 groups increased within group dispersion. Ordihull (vegan; Oksanen *et al.*, 2013) was used to plot classification groups onto nMDS ordinations.

Invertebrate community composition across all three time periods (1992, 1996 and 2017) was analysed using a non-metric multidimensional scaling (nMDS) of presence absence of species from a matrix which was extensively edited in order to match taxa across the three studies. Editing required combining species to a higher taxonomic level (usually genus). Single occurrence species were removed from this analysis. A similar process was used to analyse community composition based on abundance after dropping the 1992 data, however single occurrence species were not removed and less editing to match species across studies was required.

Results

Water chemistry

Toolibin Lake filled to a maximum depth of 2.24m in February 2017, but a breach in the diversion bank enabled some water to outflow from the wetland before it was repaired (Bourke & Rutherford, 2018). When visited in April 2017 lake depth was 1.54 m and electrical conductivity (ec) was 1072 – 1327 µS/cm with highest ec readings at the lake's southern end (Table 2). By October, lake depth had receded to 1.41 m and ec had increased to 2400- 2600 µS/cm with highest values on the lakes eastern side. In April 2017 Walbyring Lake was 0.88 m deep and salinity was lower than in Toolibin Lake (611 – 647 µS/cm). By October, salinity had increased in this lake to 1113 – 1120 µS/cm concurrent with an increase in lake depth to 0.93 m.

Salinity was lower when invertebrates and waterbirds were sampled in 2017 than at the time of any of the comparative datasets. Salinity (as total dissolved solids-TDS) at Toolibin Lake was considerably lower in both April (630 mg/l) and October (780 mg/l) 2017 than in December 1996 when Halse *et al.* (2000) recorded 9400 mg/l in conjunction with their waterbird and invertebrate sampling. Salinity at the time of waterbird surveys drawn from the Waterbirds in Nature Reserves study ranged from 460 – 10180 mg/l but most surveys were conducted at salinities closer to the mean salinity of 3090 mg/l (Unpublished SWWMP data Jim Lane). Invertebrate samples collected at Toolibin Lake by Doupe and Horwitz (1995) coincided with intermediate lake salinities of 2400-3000 mg/l.

As well as differences in salinity, there were major differences in water colour, total chlorophyll and dissolved oxygen between the 2017 and 1996 datasets. At Toolibin Lake in 2017 the water was darker with colour ranging from 500 TCU in April to 190 TCU in October, compared to 54 TCU in December 1996 (Halse *et al.*, 2000). Highest levels of colour were 700 TCU present at Walbyring in April 2017; by October 2017 these levels had declined to 440 TCU, but remained an order of magnitude higher than the 89 TCU observed in December 1996 at this lake. The high colour may have resulted from terrestrial vegetation and litter present on the lake bed prior to inundation, which would have accumulated during the previous few years of the lake being mostly dry.

In December 1996 dissolved oxygen (DO) was in excess of 200% of saturation at Toolibin Lake, a condition which typically occurs in the presence of substantial photosynthetic activity. However, chlorophyll concentrations in the water column (a surrogate for photosynthetic activity) were low (<7µg/l) suggesting that the oxygen generating photosynthesis was occurring on substrates in 1996. Similar conditions prevailed at Walbyring Lake in 1996 although DO was lower (154%) and chlorophyll concentrations in the water column were higher (36 µg/l).

The intensity of colouring substances during 2017 are likely to have reduced the amount of light available for photosynthesis, particularly on the lake bed, relative to the conditions in 1996. However, in April 2017 Toolibin Lake had substantial photosynthesis occurring in the water-column with total chlorophyll concentrations (chlorophyll a, b, c and phaeophytin) > 140 µg/l at two sites and ranged between 14 and 44 µg/l in October. Rather than elevating dissolved oxygen concentrations, vertical profiling of the water column at Toolibin Lake on 9 May 2017 indicated that DO concentrations were 15-36% of saturation at the surface and declined with increasing depth (Bourke & Rutherford, 2018). Low DO was probably in response to the high biological oxygen demand (BOD) of leaf litter and organic sediments which began decomposing when the lake flooded (e.g. Rabalais *et al.*, 2010). Negative redox potentials measured during repeat vertical profiling in December 2017 (*opp. cit*) imply that low oxygen concentrations persisted at Toolibin Lake throughout the period of waterbird and invertebrate sampling in 2017. There are no data to determine the status of DO concentrations at Walbyring Lake in 2017.

Low DO concentrations frequently result in the release of nutrients from sediments which may then promote algal blooms (Zhang *et al.*, 2010) like those which may be inferred from the high chlorophyll concentrations in Toolibin Lake during April. There was no clear evidence that nitrogen concentrations (TFN) were elevated in 2017 relative to 1996 for either wetland; although only dissolved N and P were measured (rather than total N and P which would include that assimilated by algae). However, phosphorus concentrations at Toolibin Lake in April (170 – 210 µg/l TFP) were above the recommended maximum concentration of 100µg/l (Froend *et al.*, 1997), at least 100 times greater than observed in 1996 (Halse *et al.*, 2000) and twice the concentrations observed later in October 2017 (Table 2). Whether these high concentrations were derived from lake sediments or from lake inflows directly is unknown. Concentrations of TFP in Walbyring Lake were even higher than at Toolibin in both April (530 – 610 µg/l) and October 2017 (660 µg/l). Again, it is not possible to determine the origin of this high concentration of TFP particularly in the absence of data for DO.

Under the influence of lower salinity, greater colour, lower DO and high photosynthetic activity (and presumably algal biomass) in the water column instead of the benthos, habitat conditions particularly for aquatic invertebrates were very different in 2017 compared to 1996.

Table 1. Visually estimated microhabitat characteristics at each sampling site on 27-28 October 2017.

Microhabitat (%area)	W	E	S	I	II	III
live tree	10	10	50	10	1	10
Standing dead tree	<1	<1	<1	1	0	1
Logs/stick/ bark slabs	10	25	5	10	10	30
Submerged halophytes	0	5	20	0	5	0
CPOM sediments	25	65	50	80	95	60

Table 2. Water chemistry for the three sample sites in each lake in April and October 2017.

	April						October					
Wetland	Toolibin			Walbyring			Toolibin			Walbyring		
Site	W	S	E	I	II	III	W	S	E	I	II	III
Date	11/04/2017	12/04/2017	12/04/2017	12/04/2017	12/04/2017	12/04/2017	28/10/2017	28/10/2017	27/10/2017	28/10/2017	28/10/2017	28/10/2017
Depth			1.54	0.88					1.41		0.93	
Electrical conductivity (µS/cm)	1083	1327	1072	647	611	646	1392	1483	1406	1120	1113	1120
pH	7.84	7.99	7.91	7.75	7.75	7.8	7.34	7.38	7.44	7.54	7.32	7.48
Alkalinity (mg/L)	140				138				161		116	
CO3 (mg/L)	<1				<1				<1		<1	
Ca (mg/L)	33.5				21.8				40.9		24.9	
Chloro_a (µg/L)	3.8	140	140	45	71	38	8	12	24	2	1	<1
Chloro_b (µg/L)	0.9	25	13	7.2	12	6.5	<1	<1	3	<1	<1	<1
Chloro_c (µg/L)	0.2	7.3	9.5	<0.10	<0.10	<0.10	2	2	4	1	<1	<1
Phaeo_a (µg/L)	7	28	16	96	120	130	4	4	11	4	4	4
Temperature (^{o}c)	32		26	21.3	31	19	20	26.5	14.5	18.8		17
Cl (mg/L)	265				124				390		283	
Colour (TCU)	500				700				190		440	
HCO3 (mg/L)	170				168				196		141	
K (mg/L)	13.8				16.6				15.3		13.1	
Mg (mg/L)	24				14				28.7		19.3	
N_NO3 (mg/L)	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01		0.15	
N_totsol (mg/L)	5	3	3.6	5.5	5.1	5.8	2.4	2.5	2.6		3.9	
Na (mg/L)	132				80.2				196		167	
P_totsol (mg/L)	0.21	0.16	0.17	0.53	0.54	0.61	0.087	0.069	0.1		0.66	
SO4 (mg/L)	4.3				1.7				7.8		22.6	
Turbidity (NTU)	3.9				14				6.9		9.7	
TDS (mg/L)	630				520				780		610	

Waterbirds

During 2017 19 species of waterbird were recorded for Toolibin Lake and 16 species were recorded at Walbyring Lake. All species had been recorded at the wetlands previously. At Toolibin Lake the April survey had lower richness than the October survey, but at Walbyring the surveys were similar.

Species richness of individual waterbird surveys was similar across all four time periods at Toolibin Lake (Fig. 1). During the 1980s, richness varied between 1 and 23 species with a median richness of 17. This variability reflected the wide range of depth (0.1 – 2.5 m) and salinity (0.95 – 4.8 g/l) at which surveys were conducted, however there was no statistically significant correlation between richness and either variable. The four spring “pre-salinity baseline” surveys from the 1980s (Froend *et al.*, 1997) had richness in excess (i.e. 18-22 species) of the median, while Froend’s autumn pre-salinity example had lower than median richness (15 species). The autumn 2017 survey also had lower richness (13) than was recorded in the spring survey (17) of the same year. With 19 species recorded in 1975 and again in 1996 the available surveys do not suggest any change in species richness at Toolibin Lake across the four time periods. Walbyring Lake had lower species richness than Toolibin Lake on all occasions. Richness at Walbyring Lake during 1996 and 2017 was close to the maximum observed during the 1980s at this lake.

Eight feeding guilds were defined by Halse (1987) to describe the bird community at Toolibin Lake. The number of species (richness) in each of these guilds varied across the 4 time periods (Fig. 2) indicating some changes in the functional arrangement of the waterbird community over time.

Dabblers was the most species rich guild in all time periods; with richness approaching the total membership of nine species recorded at Toolibin Lake in all previous surveys (i.e. Munro, 1975; Goodsell *et al.*, 1978; Jaensch *et al.*, 1988; Halse *et al.*, 2000). Richness of the guild varied across the large number of individual surveys in the 1980s; suggesting a dependency on prevailing conditions and reflecting the overall richness of waterbirds. However, median richness in the 1980s was 7 species and 50% of surveys had between 6 and 8 species. In 1975 and 1996 richness of the dabbler guild was higher than the median, but within the interquartile range of 1980s data whereas richness was always slightly lower than the 1980s median in 2017.

Two guilds, divers with an animal diet (cormorants and grebes) and large waders (herons, egrets and spoonbills) had richness during 2017 at the lower end of that observed during the 1980s (Fig. 2). Divers generally prefer depths > 1 m (Halse, 1987) and low richness of the guild during the 1970s and 1990s, when lake depth was 0.76 and 0.66m respectively, can be attributed to insufficient depth during these individual surveys (all members of the guild were included in the species list of Goodsell *et al.*, 1978). However, during 2017 depth was > 1.4 m, comparable to the depth of many of the 1980s surveys and unlikely to have restricted diver richness, suggesting other factors, such as food availability and factors affecting colonisation (e.g. regional availability of colonists or availability of alternative suitable wetlands), were important. The cormorants (little black, little pied, pied and great) none of which were present in 2017 might be expected to be influenced by food availability and forced to seek alternative wetlands where food was insufficient. These are the larger species of the guild and require more and larger food items than the grebes. In contrast, the hoary headed grebe (the smallest and only member of the guild occurring in all time periods) not only has a less size dependent diet but also higher rates of occurrence regionally and occurs in a wider range of inland wetland types than the cormorants (e.g., compare waterbird group 2 and 1 of Halse *et al.*, 1993). Consequently, this species will more readily find, colonise and prosper in a newly filled wetland such as Toolibin Lake.

The large wader guild (herons etc.) was consistently represented by 4 of the 5 member species prior to the 2017 surveys. In 2017 the guild was represented by just the white-faced heron (12 individuals in April and 13 in October) and white-necked heron (2 individuals in October). The former, like hoary-headed grebe, is a tolerant, widely occurring species adding further support to the view that observed patterns of low richness amongst the diver-animal and large wader guilds in 2017 may reflect a lower quality of lake habitats regionally, suitable only for tolerant species, compared to previous time periods.

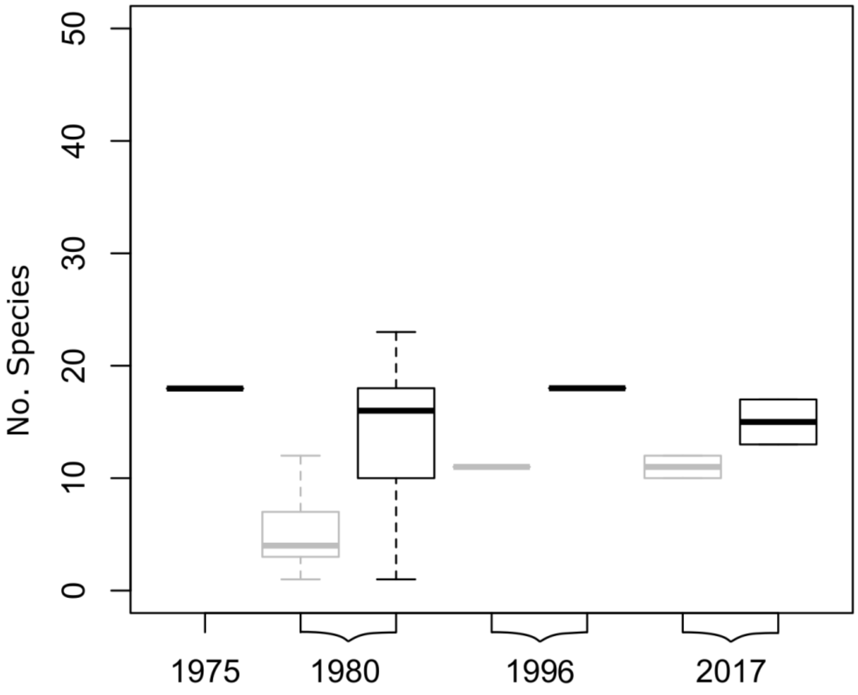

Figure 1. Waterbird species richness for individual surveys at Lake Walbyring (grey) and Toolibin (black) in four time periods. Box plot comprises; median (horizontal bar), inter quartile range (box) and whiskers to minimum and maximum value. 1975 data collected by Munro (1975), 1980s data from Jaensch et al. (1988). The 1990 data was collected in 1996 (Halse *et al.*, 2000).

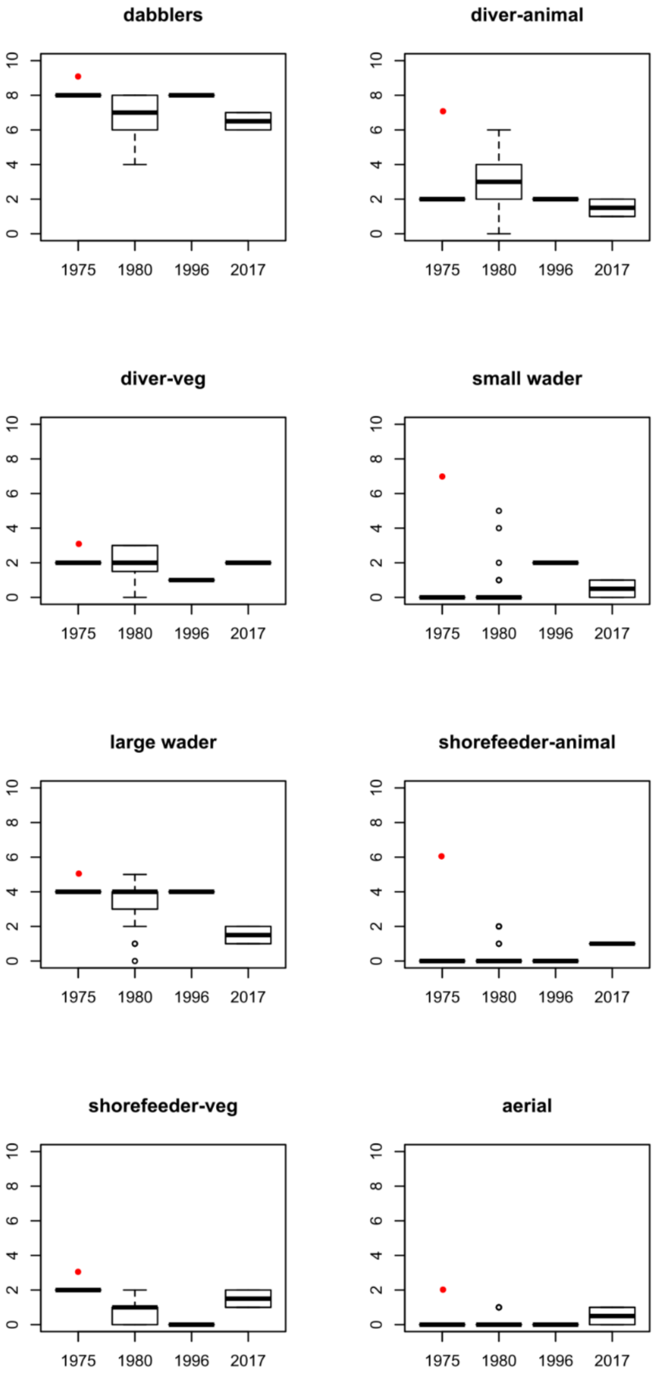

Figure 2. Box plots of the richness of eight water bird guilds (after Halse, 1987) at Toolibin Lake. Box plots comprise median (horizontal bar), inter quartile range (box), whiskers to greatest value within 1.5 x the inter quartile range and an open circle representing extreme values. Data sets as in Figure 1 with the exception that only 1980s surveys with depth > 1.4 m were included and with the addition of a filled circle representing the total richness of the guild in the cumulative species list for 1968-75 (Goodsell *et al.*, 1978).

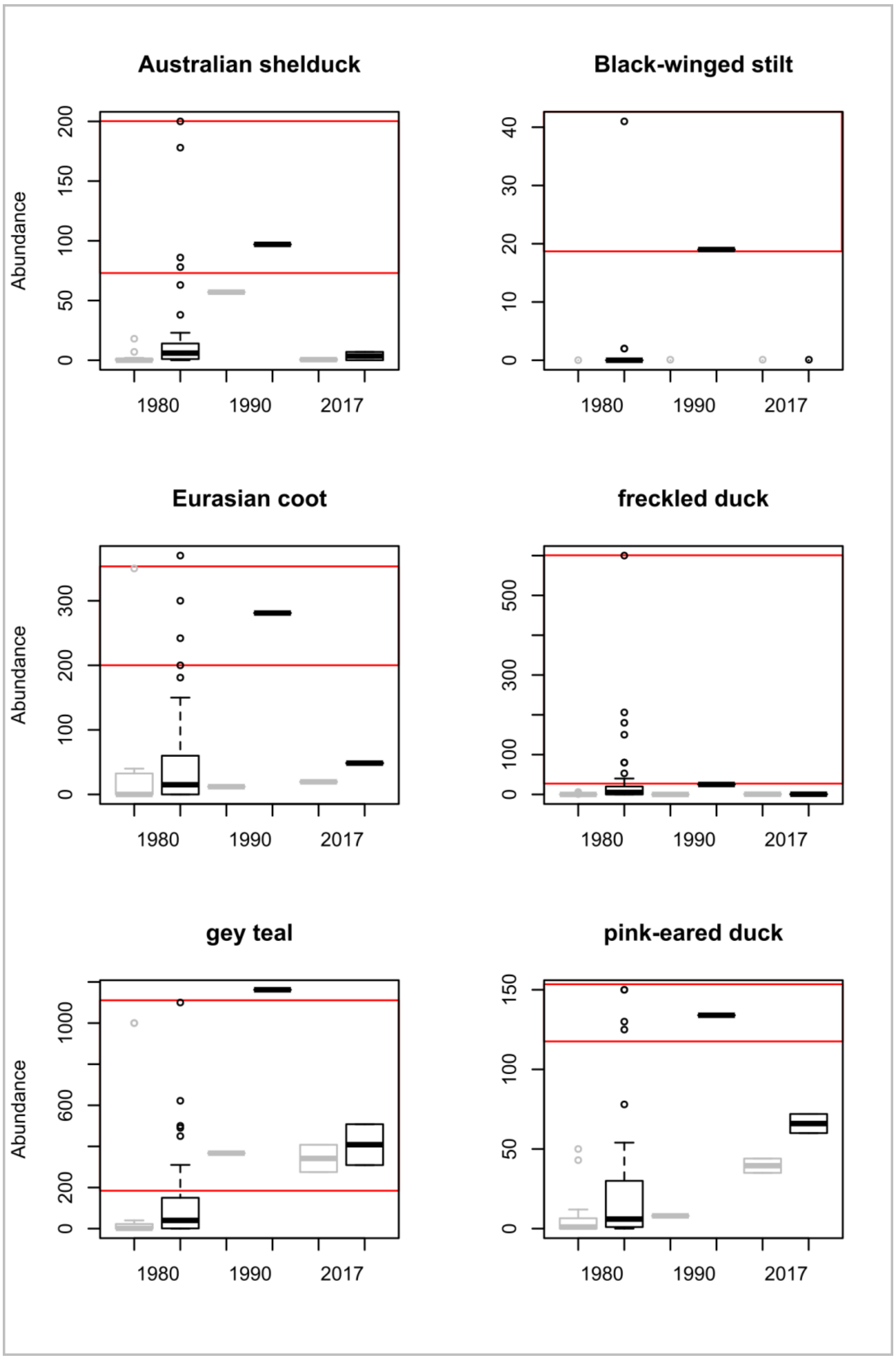

Figure 3. Total abundance during three time periods of the 6 indicator bird species listed in the Toolibin recovery plan (DBCA, 2017). Box plots comprise median (horizontal bar), inter quartile range (box), whiskers to greatest value within 1.5 x the inter quartile range from the median and open circle representing extreme values; Toolibin Lake in black, Walbyring Lake in grey and red boundary lines represent the target abundance range for each species.

Other feeding guilds i.e., small waders, shore feeders and aerial feeders had low richness, few members and/or occurred infrequently across all time periods. They are difficult to draw conclusions from, but there is no evidence that their occurrence in 2017 was comparatively low. While the small wader guild had many member species they were largely absent from the analysis because the high richness surveys investigated occurred at greater lake depths than preferred by this guild.

The recovery plan for Toolibin Lake (DBCA, 2017) documents six indicator waterbird species; Australian shelduck, pink-eared duck, grey teal, freckled duck, Eurasian coot and black-winged stilt. An aim of the recovery plan is to maintain the presence of breeding populations of these species in the lake at abundances similar to those in earlier surveys. All indicator species were present at Toolibin Lake in 1996 and, with the exception of the black-winged stilt, again in 2017. The indicator species were also present at Walbyring Lake in both 1996 and 2017, except for black-winged stilt. Black-winged stilt have a relatively low frequency of occurrence in available surveys from Toolibin Lake. They were recorded in December 1996 and in three surveys during 1983 all at lake depths of < 0.7m; suggesting the lake was too deep for this species during the survey periods in 2017. Black-winged stilt were present in the 1970s cumulative species list but not in the single 1975 survey at a depth of 0.76m. To assess the presence and abundance of this species against management goals requires waterbird surveys to be conducted at lower depths later in the hydrological cycle.

In 2017, the abundance of Australian shelduck, Eurasian coot and freckled duck were within the interquartile range of abundances observed in the 1980s, but below the management target abundances for Toolibin Lake. The freckled duck was recorded as single individuals in both lakes during October 2017; an abundance lower than the median of 2 and 13 individuals, at Walbyring and Toolibin respectively, during the 1980s, which is in turn lower than the abundance target range of 25-600 individuals (DBCA, 2017). Only a small number of the surveys from the 1980s and the 1996 survey included freckled duck with abundance within this broad target range. This was also true for Australian shelduck and Eurasian coot, suggesting that the target ranges may be too high for single surveys to assess. These abundances are likely to occur more frequently in the recessional drying phase of the wetland when birds are aggregating because of diminishing availability of water elsewhere. For example, abundances for the indicator species were relatively high during 1996 (Fig. 3) and reflect what was the recessional stage of that hydrological cycle. This contrasts, however, with the few surveys during the 1980s meeting the target abundances which were from various stages of the hydrological cycle and more likely reflect genuinely optimal periods for waterbirds at Toolibin Lake.

Only grey teal occurred with abundances in the target range during 2017. The abundance of grey teal and pink-eared duck was higher than the median (and interquartile range) at both lakes compared to the 1980s, but less than the maximum abundances recorded. Pink-eared duck were below the target abundance despite having greater abundance than 75% of surveys during the 1980s.

During 2017 there was evidence of breeding for six species. Broods were detected at Toolibin Lake for pink-eared duck (April and October), grey teal (April) and Eurasian coot (April), but not for the remaining three indicator species. Other species breeding at Toolibin lake were hoary-headed grebe (April) and Australian wood duck (April). At Walbyring Lake, broods were detected for grey teal (April and October), pink-eared duck (April and October) and musk duck (October). The five species recorded breeding at Toolibin Lake were also recorded breeding during the 1980s when a total of 22 species, including all indicator species except the black-winged stilt, were recorded breeding (Jaensch *et al.*, 1988). Numerous species that were recorded breeding during the 1980s, including species of cormorant (3), heron (3), and egret (1), were not present during 2017 surveys. However, three species, Australian shelduck, white-faced heron and black swan which bred regularly during the 1980s were present but not observed breeding during 2017, raising the question of whether conditions were un-suitable or perhaps less suitable than elsewhere. Five species breeding in April 2017 is comparable to the 1-6 breeding species (mean 3.6) reported by Froend et al. (1997) for surveys considered suitable as a pre-salinity baseline.

Species richness and the small differences in richness of individual feeding guilds were the main changes in community composition across the four time periods. Consequently, an ordination of the community composition of all available surveys (Fig. 4) separates surveys where low richness was recorded (groups 1-3 and 5) from the bulk of surveys (group 4) that includes the majority of surveys from the 1980s (particularly those from periods of greatest depth) and the 1996 and 2017 surveys. The 2017 and 1996 surveys are both placed to the left hand side of group 4 while the “pre-salinity” baseline surveys described by Froend *et al.* (1997) lie to the right of the group, describing a continuum of community composition across which the majority of surveys from the 1980s are arranged. This continuum is strongly defined by the low richness of diver and large wader guilds to the left (i.e. 2017 and 1996 surveys) and also by the dominance of grey teal which make a relatively greater contribution to total abundance in these surveys. While this may represent the effects of a change in habitat suitability since the 1980s, this is not unequivocal given the significant number of surveys from the 1980s with similar community composition.

In summary, the waterbird community in 2017 at Toolibin Lake had similar species richness and total abundance to some surveys from earlier time periods. However, there is evidence that several species have been lost from the community since as early as 1996. Key species such as the freckled duck continue to be present, but most recently were of very low abundance. The number of breeding species in 2017 surveys were similar to that recorded in the pre-salinity baseline surveys (Froend *et al.*, 1997).

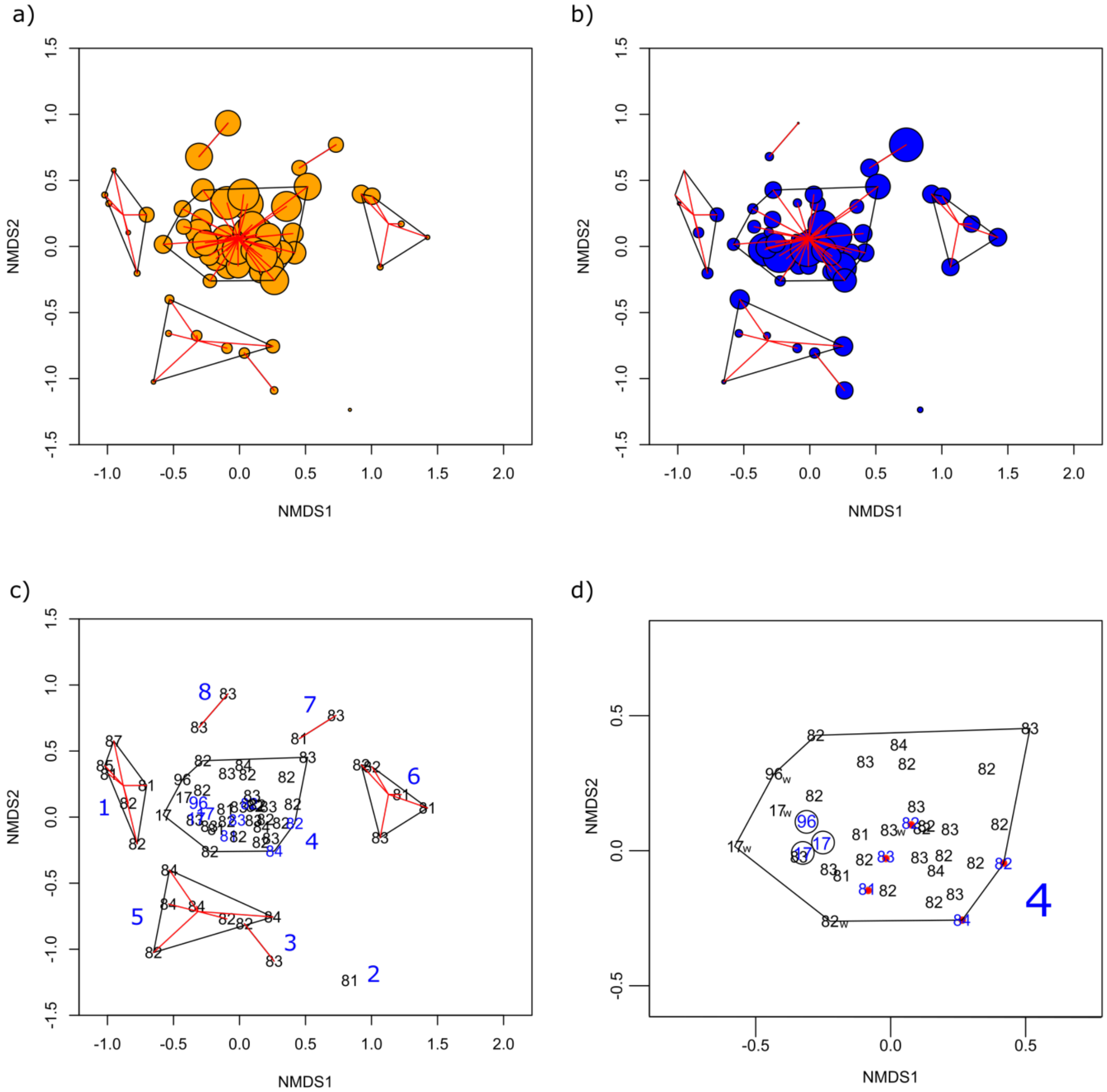

Figure 4. NMDS ordination of waterbird community composition (species abundance) at lakes Toolibin and Walbyring (stress=0.188). Enclosing polygons encompass groups determined by a UPGMA classification of the community data. Survey points are scaled by a) species richness and b) lake depth. c) Surveys labelled by sampling year and d) detail of group 4 with contemporary Toolibin surveys circled and pre-salinity baseline surveys (Froend *et al.*, 1997) marked by dots.

Aquatic invertebrates

Toolibin vs Walbyring

Historically, Walbyring Lake has supported an aquatic invertebrate community of greater species richness than Toolibin Lake (Doupé & Horwitz, 1995; Halse *et al.*, 2000). However, richness was greater in Toolibin Lake in April and October 2017. Individual sample richness was 12-25 in April and 38-43 in October at Toolibin Lake compared to 11-19 and 27-37 species for the same periods at Walbyring Lake (Fig. 5). Similarly, the total number of species collected at each wetland in 2017 was higher at Toolibin (35 and 57 for April and October respectively) than at Walbyring (22 and 42 species).

Community composition at Walbyring Lake has also, historically, been distinct from that of Toolibin Lake. More site specific species were collected from Walbyring Lake in 1992 (Doupé & Horwitz, 1995) and an ordination of either presence/absence or abundance of species clearly indicates a distinct composition in each wetland in 1996 (Fig. 6 and see Halse *et al.*, 2000). This is in contrast with very similar community composition in the two wetlands in 2017 (Fig. 6).

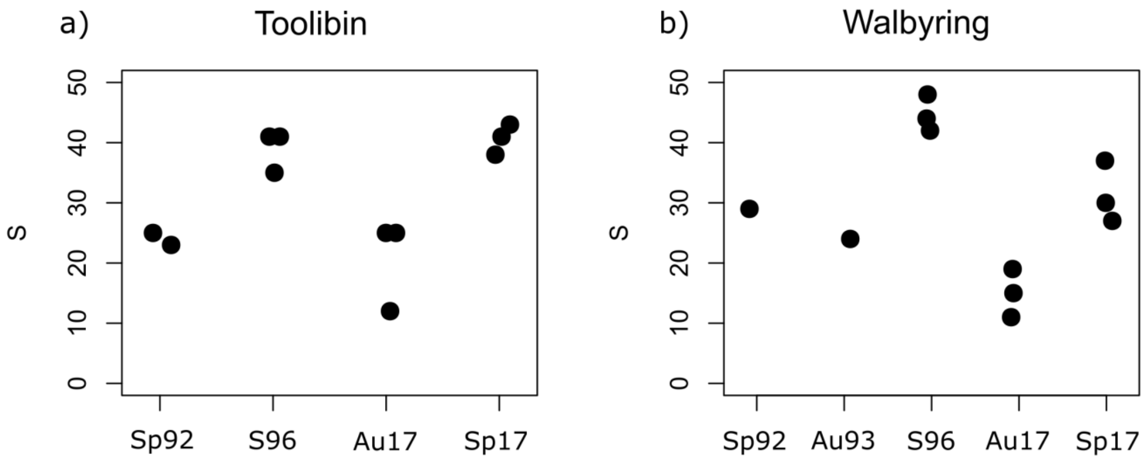

Figure 5 Invertebrate species richness (each sample) from 3 studies at lakes a) Toolibin and b) Walbyring. Sp92 and Au93 are spring and autumn samples in 1992 and 1993 (Doupé & Horwitz, 1995), S96: summer 1996 (Halse *et al.*, 2000) and Au17 and Sp17: autumn and spring 2017. Sample points in each time period are slightly 'jittered' for clarity.

Higher richness at Toolibin compared to Walbyring during 2017 may be the result of the smaller difference in salinity between the two wetlands. During the 2017 fill event, salinity across the two sampling events ranged from 630-780 mg/l at Toolibin and 520 – 610 mg/l at Walbyring (Table 2). Salinity in Toolibin Lake was 9400 mg/l in December 1996; slightly more than three times the salinity in Walbyring Lake at the same time. Similarly, in spring 1992 salinity at Toolibin Lake (2400 mg/l minimum) was twice that in Walbyring Lake (Table 1 Doupé & Horwitz, 1995). Pinder et al. (2005) showed that salinity above 4000 mg/l (or 2600 mg/l for freshwater species only) constrained species richness in other Wheatbelt wetlands. In 1996, salinity in the two wetlands straddled this constraining threshold and probably depressed the richness of invertebrate communities in Toolibin Lake. A smaller difference in richness between the lakes in 1992 (Fig. 5) reflects both the lower salinity and the smaller difference in salinity between the two wetlands. In 2017, richness of both wetlands was probably unconstrained by the prevailing salinity which was less than the 2600mg/l below which Pinder et al. (2005) could find no effect of salinity on richness. When coupled with the larger area and greater habitat heterogeneity (provided by more live trees, sticks, logs and emergent halophytes; Table 1) it is unsurprising that richness would be greater in Toolibin Lake when salinities are similar in both wetlands. Given similar salinities, the conservation value of the invertebrate communities of both wetlands are probably similar, but because the species present are generally widely distributed the communities are likely to have greater importance in support of the waterbird community than for the suite of species present (e.g. Halse *et al.*, 2000).

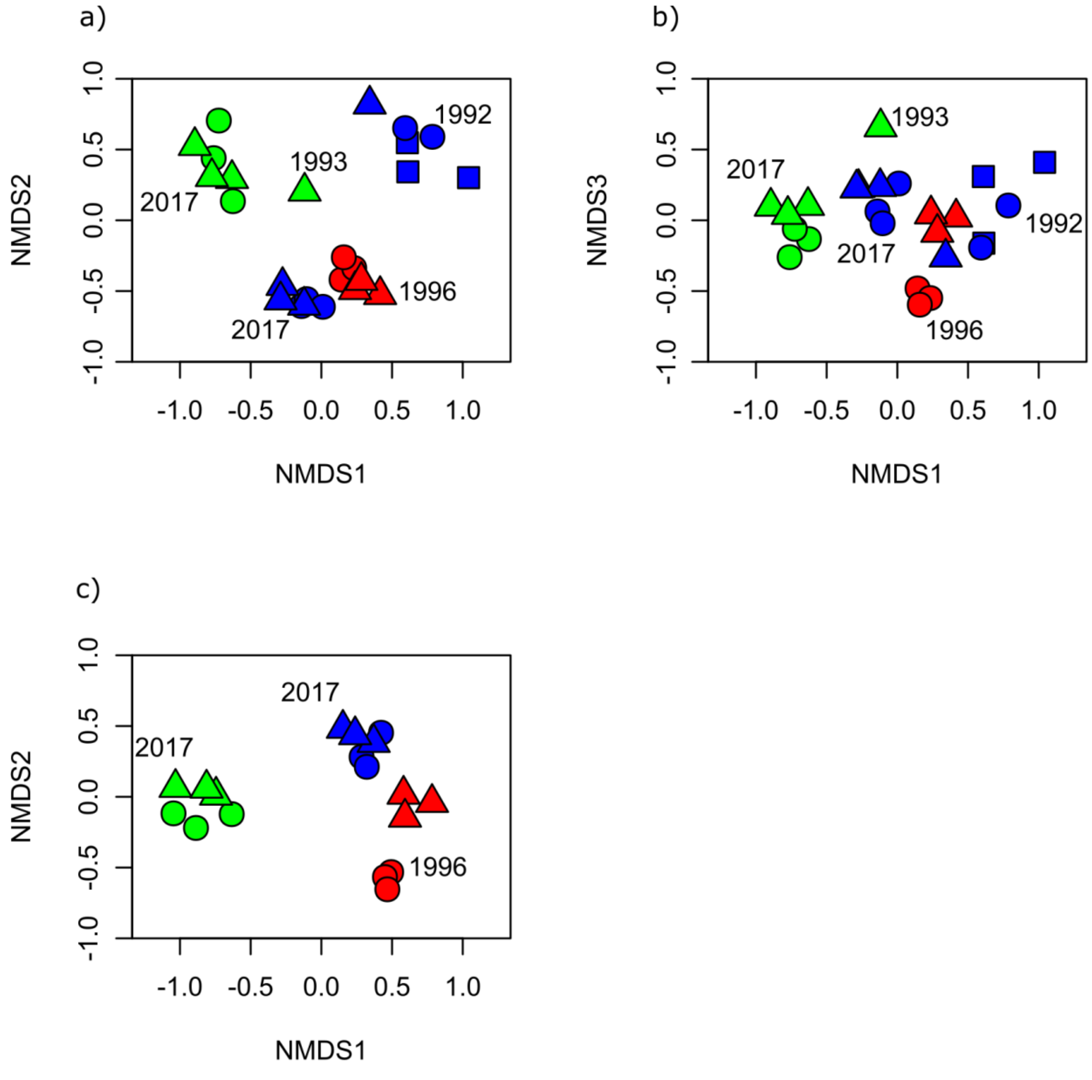

Figure 6 Two ordinations (NMDS) of invertebrate community composition at Toolibin and Walbyring Lakes; a-b) the axes of a three dimension ordination (stress = 0.069) of species presence/absence, from 1992 and 1993 (Doupé & Horwitz, 1995), summer 1996 (Halse *et al.*, 2000) and autumn (April) and spring (October) 2017 surveys, c) a two dimension ordination (stress = 0.042) of species abundance from the same 1996 and 2017 surveys. Seasons are colour coded; green=autumn, blue=spring, red=summer. Toolibin is represented by circles, Walbyring by triangles and other sites in the catchment by squares.

Low April Richness

The total richness of aquatic invertebrates (across 3 samples) at Toolibin Lake was lower in April 2017 (43 species) than in October 2017 (68 species). This was reflected in the richness of individual samples which were as low or lower during April at both wetlands than on any other sampling occasion (Fig. 5).

Low richness at Toolibin in April 2017 probably coincided with low concentrations of dissolved oxygen (DO) in response to the high biological oxygen demand (BOD) of large volumes of organic material accumulated since the last significant inundation of the wetland in 2006. Because of its effect on DO, large volumes of organic matter may depress species richness and impede community development among invertebrates in recently filled wetlands in the short term despite being the basis of food chains in the longer term (Boulton *et al.*, 2014).

In addition to reduced richness the effects of low DO on the invertebrate fauna were relatively clear from the types of species present in April, which formed a community compositionally distinct from other sampling occasions (Fig.6). In the benthic zone, where DO was lowest, there was a predominance of species tolerant of low DO such as the midges *Chironomus alternans* and *Chironomus occidentalis*. These species are able to tolerate waters with considerable organic pollution and consequent low DO (Johnson, Wiederholm & Rosenberg, 1993). Additionally, the water column was dominated by a very high abundance of cladoceran filter feeders of the genus *Moina*; some species of which have considerable tolerance of low DO (e.g. Svetlichny & Hubareva, 2002; Leung, 2009).

There is some evidence that low DO persisted until after the October 2017 sampling occasion, however a persistent effect on invertebrate community structure was less apparent. While low DO tolerant *Chironomus* species were still present in October their abundance was lower and many chironomid species with less tolerance of low-oxygen, such as *Dicrotendipes*, *Polypedilum* and *Tanytarsus* (Johnson *et al.*, 1993 and references therein) were more abundant. Similarly, the general increase in species richness at both wetlands in October would be unlikely if DO remained low and was structuring the community. It is possible that DO concentrations between May and December (i.e. between the measurements of Bourke et al.(2018)) were not low enough to alter community structure. It is also possible that some other factor (for example low rates of colonisation and egg hatching) was responsible for the lower richness in April relative to October.

Differences in fauna over time

In 1992, the richness of insects (56% of the fauna) and crustaceans (44%) was similar at Lake Toolibin. The relative proportions of these taxa at Walbyring Lake displayed a very similar pattern. In summer 1996, the fauna was more insect dominated (69%), with crustaceans only 30% of richness. In April 2017, the invertebrate fauna (at Toolibin) was even more dominated by insects (78%) with crustaceans accounting for just 14%. In October, the proportions of richness comprised of insects declined to (65%) and crustaceans increased to 24%.

The large proportion of insects in April 2017 indicates that most colonising species were probably coming from other wetlands and dependent on the dispersal of winged adults. The increasing proportion of crustaceans in October suggests that the hatching of propagules originating in the lake and/or arriving on waterbirds (e.g. Green *et al.*, 2008) increased in importance.

Seventeen species of Rotifera were recorded from April 2017 samples. This assemblage of micro-invertebrates was dominated (both occurrence and abundance) by *Filinia* sp., *Polyarthra* sp. and several species of *Brachionus* (Appendix 2). These genera have very short generation times, are frequently encountered in episodically filling wetlands and, depending on species, are salt tolerant. While the presence of rotifers was noted in 1996 (Halse et al 2000) their diversity was not determined.

During the 1970s a freshwater fauna at Toolibin Lake was indicated by the observed presence of 2 species of fish, tadpoles, aquatic insects, bivalves, snails, shrimps and gilgies (Goodsell *et al.*, 1978) at a time when salinity was 1-2 g/l (approx.1600 – 3200 µS/cm). It has been suggested that the fish species were probably *Pseudogobius olorum* and *Atherinisoma wallacei* (Froend & Storey, 1996). Similarly, these authors suggested that the bivalve may have been the now threatened *Westralunio carteri*, the shrimp, *Palaemonetes australis*, and the gilgie, the introduced *Cherax destructor*. These species were probably reliant on regular colonisation into the Toolibin wetland chain from source populations in the Arthur River. As far as I am aware aquatic insects and snails are the only groups on the list of Goodsell et al. (1978) that have been recorded in Toolibin Lake since the 1970s.

The invertebrate fauna in 1992 (Doupé & Horwitz, 1995) and 1996 (Halse *et al.*, 2000) show a progressively more salt-tolerant community, although part of this progression is the result of sampling later in the hydrological cycle. Despite low salinity in 2017, the overall fauna was still a salt tolerant one which may reflect the egg bank (which would last have been replenished when the lake was more saline) and the generally saline nature of other wetlands from which other colonists could have originated.

There were three salt-intolerant species recorded in 2017. *Hydra* sp and *Simocephalus gibbosus* are typical of freshwater and were present at Toolibin in October 2017. Another typically freshwater species was the oligochaete *Dero digitata* which was present in both wetlands in October 2017. The absence of these species in April 2017 suggests a period of time may have been required for colonisation from elsewhere probably with waterbirds as the vector (e.g. Green *et al.*, 2008). None of these species, or any other freshwater dependent species, were present in 1996. Oligochaeta were not identified to species in 1992 so one of the oligochaetes identified by Doupe and Horwitz at that time may have been a *Dero.* Several other typically freshwater species found in 1992 (Appendix 1; Doupé & Horwitz, 1995) included the gastropod *Physastra* sp. at Walbyring Lake and the decapod shrimp *Paleomonetes australis* in three sites upstream of Toolibin Lake.

At Toolibin invertebrate richness in October 2017 was similar to that observed in December 1996 despite the presumed constraint of higher salinities in the latter sampling period. The mean richness of invertebrate samples in October 2017 was 40.6 and lower than expected given a mean of 60 3 (± 17.9) species observed at forty-nine Wheatbelt wetlands with similar salinity (<1000 µS/cm) (data from Appendix 3; Pinder *et al.*, 2009). This contrasts with the situation in 1996 where the mean richness of 39.3 was comparable to the 43.1 (± 9.4) observed for 10 wetlands with comparable salinity (i.e. 15000-20000 µS/cm) (data from the same source). The lower than expected richness in October 2017 may in part be due to low DO as discussed above. However, several features of the invertebrate community including the low richness of ostracods and salt-intolerant species and the high proportion of highly dispersive insects suggest that sources of colonists were compromised for many species.

Boulton and Lloyd (1992) showed that resting egg banks declined in viability as time since inundation increased. Egg banks of resting stages (of invertebrates such as ostracods and cladocerans) within the wetlands' sediments may have been depleted since the late 1990s if the minor inundation events over the intervening 20 years were not sufficient to replenish them. It could also be the case that the condition of neighbouring wetlands has declined and now support fewer potential colonising species. The entire suite of rotifer species was found in both Toolibin Lake and Walbyring Lake eight weeks after filling, suggesting that they hatched from local sediments. It is likely that resting stages of these short-lived species are replenished even during short periods of inundation.

Invertebrates as waterbird food

Chironomids were an important component of the invertebrate fauna at both wetlands in 2017 and showed a succession of species from *Chironomus occidentalis* and *C. alternans* at high abundance (10000 and 1000 individuals per sample respectively) in April to a more diverse mix including *P. nubifer*, *Kiefferulus intertinctus*, *Procladius paludicola*, *P. villosimanus*, *T. fuscithorax* and *D. conjunctus* at moderate to high abundance in October. This succession matched closely that observed in two recently flooded wetlands in New South Wales (Maher & Carpenter, 1984) except that *Chironomus tepperi* and *C. alternans* were the early colonists. *C. tepperi* was the only species recorded at Toolibin in September 1992 (Doupé & Horwitz, 1995), but the chironomid assemblage in December 1996, while less abundant, was very similar to the assemblage in October 2017; assuming that *Chironomus oppositus* recorded in 1996 is the same as *C. alternans* recorded in 2017 (e.g. Martin, 2013) and noting that that *D jobetus* in 1996 was replaced by *D. conjunctus* in 2017.

Both adults and larvae of *C. tepperi*, can be an important food source for a number of duck species including Pacific black duck, grey teal, pink-eared duck and freckled duck, and a correlative link between the intensity of waterbird breeding and the abundance of this species 8 weeks earlier has been reported (Maher & Carpenter, 1984). The larvae of *C. alternans, C. occidentalis* and *K. intertinctus* are large and of similar size to *C. tepperi* and it is unlikely that they are not also important food sources taken in proportion to their abundance. In October 2017, chironomids were probably the most important prey species for waterbirds because of their size and abundance. This was probably also the case in April 2017, however abundance was much lower and juvenile hemipterans of the genera *Agraptocorixa*, *Sigara* and *Anisops* made a significant contribution to available prey species.

In December 1996, ostracods were probably a more important food item for waterbirds than were chironomids which, despite a diverse assemblage of species, were of low abundance. At this time 6 species of ostracod were recorded for Toolibin Lake (the original species list has 7 but individuals of *Mytilocypris mytiloides* and *Mytilocypris tasmanica chapmani* are now all considered to be *M. mytiloides*) and were collectively very abundant. The relatively large ostracods; *M. mytiloides*, *M. ambiguosa*, *Alboa wooroa* and *Bennelongia australis* were likely to have been the most significant prey species at this time and indicated very different food availability for waterbirds. A similarly diverse assemblage of ostracods was present in 1992, comprising five of the 6 species present in 1996 plus one additional species (Doupé & Horwitz, 1995). Salinity was lower at this time but still higher than during 2017.

Ostracods were absent from Toolibin Lake in April 2017 and represented by only 2 species (*Alboa wooroa* and *Sarscyprodopsis aculeata*) in October. A study of the Drummond wetlands

(Pinder et al. 2014) suggests that the eight weeks between first inundation and the April sampling is plenty of time for a diverse ostracod fauna to have developed and this would certainly have been the case by October (36 weeks after filling, which is greater than the 24 weeks between the June 1996 filling and December 1996 sampling). The absence of ostracods in April 2017 may be due to the low oxygen concentrations but it is also possible that the drought resistant egg bank in the sediments had become depleted during the many years prior to 2017 without a significant filling event. As mentioned above, crustacean egg banks become less viable as sediments are flooded less frequently (Boulton and Lloyd 1992). Three of the six species present in 1996 are halophytic (*Diacypris spinosa*, *M. mytiloides* and *M. ambiguosa*) and their absence in 2017 may be related to the lower salinity, but it would have been expected that these would be replaced with at least a similar number of freshwater species.

Large invertebrates such *Cherax destructor* and *Paleomonetes australis*, in conjunction with fish and tadpoles of *Helioporus* sp. and *Lymnodynastes dorsalis* would have been an important part of the diet of waterbirds at Toolibin and Walbyring lakes. *Limnodynastes dorsalis* was heard calling at Toolibin Lake in April 2017 but no tadpoles of any species were observed. Given the general salinisation of the catchment the decline of these species is likely to be widespread, limiting the opportunity for their recolonization of Toolibin or Walbyring Lakes when filled. The absence of these historic food species may largely explain the decline in richness of both the diver and large wader waterbird guilds since 1996.

Conclusions

Current Status of Toolibin Lake Fauna

Invertebrate community structure at Toolibin and Walbyring Lakes was similar during 2017 although species richness was relatively higher at Toolibin Lake, a feature that has not previously been reported. Community structure was likely to have been influenced by a mix of both environmental filtering, as low DO depressed richness and altered composition particularly in April, and colonisation effects, with some evidence that the availability of colonising species was reduced, possibly by factors such as depleted egg banks and lower habitat quality in neighbouring wetlands. These structuring effects were greater in April, shortly after filling, than in October; which may suggest that a sufficient hydroperiod would see a continued development of the communities. However, compared to healthy wetlands such as those of the Drummond Nature Reserve their rate of development appears to have been retarded. If long periods without inundation and shorter hydroperiods continue to be the norm at Toolibin Lake, it is likely that invertebrate communities will not reach their full potential without measures to improve recolonisation when the lake fills. Given that improvement of other natural wetlands in the catchment is unlikely to precede improvement at Toolibin or Walbyring lakes, such measures might include linking artificial freshwater wetlands (such as farm dams) to the inflow of Toolibin Lake.

Despite invertebrate community composition falling short of what might be expected given the improved water quality during 2017, there is no evidence that this restricted the development of water bird communities. Waterbird species or groups absent during 2017 became less common between the 1980s and 1996. The abundance of invertebrate species typically preyed on by waterbirds was lower in 2017 than during 1996 but included mostly larger species so it is unclear how this would have compared in terms of waterbird energetics. The breeding of 6 waterbird species during 2017 is comparable with that reported for pre-salinity surveys selected by Froend *et al*. (1996), So, food availability did not affect the richness of breeding species although there is insufficient data to determine if the number of raised broods has declined. Similarly, it is unclear whether available food resources limited waterbird abundance during 2017 relative to other years. The abundance of all but one waterbird indicator species was below the desired management target range, however, species abundances are affected by other factors in addition to food resources including other aspects of the quality of habitat and the extent of water in other wetlands. Many surveys during the 1980s supported abundances similar to those encountered in 2017 so it is unclear whether a failure to meet abundance targets on any single occasion is due to a lack of food resources or other periodically suboptimal conditions. There are published techniques available for determining the number of waterbirds that can be supported by a wetland's resources (Heitmeyer, 2010) but the data required, such as invertebrate biomass and species density estimates, are not currently available and might be considered when developing research to fill knowledge gaps in the future.

The absence of a number of large prey species such as *Cherax*, tadpoles and fish since at least the 1996 survey is likely to have resulted in lower suitability of the wetlands for diver and large wader species, which had low diversity in 2017 and represents the single biggest change in community structure of waterbirds since the 1980s. The loss of reed habitats in the 1970s also caused a change in community structure with the loss of Australasian bittern and purple swamphen (Froend *et al.*, 1997). On a positive note the management of Toolibin Lake's hydrology has preserved stands of live trees (DBCA, 2017) which are very important to the diversity of waterbird species breeding in the lake (Halse, 1987) and the loss of which would almost certainly result in major changes to the waterbird community.

Acknowledgements

Maria Lee (Wheatbelt Region, DBCA) assisted the authors with the April 2017 field work. Dr Russ Shiel identified and calculated the relative abundance of micro-invertebrates. Jasmine Rutherford and Lindsay Bourke are thanked for access to reports while in preparation and for sharing their knowledge of hydrology. Russel Shiel (University of Adelaide) identified the rotifers and cladocera. Yvonne Winchcombe and the SWWMP program team are thanked for Toolibin Lake data. Funding assistance for field work was provided by Greg Durrell DBCA Regional Manager, Wheatbelt Region.

Bibliography

Boulton A.J., Brock M.A., Robson B.J., Ryder D.S., Chambers J.M. & Davis J.A. (2014). *Australian Freshwater Ecology: Processes and Management*, 2nd edn. John Wiley & Sons, Ltd, South Australia.

Bourke L. & Rutherford J. (2018). *Hydrological response of Toolibin Lake to inundation in February 2017*. Department of Biodiversity, Conservation and Attractions, Kensington, Western Australia.

Bowra T. & Wallace K.J. (1996). *Toolibin Lake recovery Team Annual report 1996*. Departmant of Conservation and Land Management.

Casson N.E. (1988). *Lake Toolibin and Reserves Interim Management Guidelines*. Department of Conservation and Land Management.

DBCA (2017). *Lake Toolibin Catchment Recovery Plan 2015-2035*. Department of Biodiversity, Conservation and Attractions, Perth.

Doupé R.G. & Horwitz P. (1995). The value of macroinvertebrate assemblages for determining priorities in wetland rehabilitation: A case study from Lake Toolibin, Western Australia. *Journal of the Royal Society of Western Australia* **78**, 33–38

Frith H.J. (1959). The ecology of wild ducks in inland New South Wales. IV. Breeding. *Wildlife Research* **4**, 156–181

Froend R.H., Halse S.A. & Storey A.W. (1997). Planning for the recovery of Lake Toolibin, Western Australia. *Wetland Ecology and Management* **5**, 73–85

Froend R.H., Heddle E.M., Bell D.T. & McComb A.J. (1987). Effects of salinity and waterlogging on the vegetation of Lake Toolibin, Western Australia. *Australian Journal of Ecology* **12**, 281–298. https://doi.org/10.1111/j.1442-9993.1987.tb00949.x

Froend R.H. & Storey A.W. (1996). *Monitoring Design and Data Analysis Toolibin Lake and Catchment Part 1: Review and Analysis of Monitoring Data*. Department of Conservation and Land Management, Western Australia.

Goodsell J.T., Garstone R. & Lambert P. (1978). Guidlines to the management of the fauna of Lake Toolibin nature reserve and other nearby nature reserves. In: *Northern Arthur River wetlands Rehabilitation Committee: progress report, August 1978*.

Green A.J., Jenkins K.M., Bell D., Morris P.J. & Kingsford R.T. (2008). The potential role of waterbirds in dispersing invertebrates and plants in arid Australia. *Freshwater Biology* **53**, 380–392

Halse S.A. (1987). *Probable effect of increased salinity on the waterbirds of Lake Toolibin*. Department of Conservation and Land Management, Perth.

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

Halse S.A., Pearson G.B., McRae J.M. & Shiel R.J. (2000). Monitoring aquatic invertebrates and waterbirds at Toolibin and Walbyring Lakes in the Western Australian wheatbelt. *Journal of the Royal Society of Western Australia* **83**, 1–17

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

Hamilton-Brown S. & Blyth J. (2000). *Perched Wetlands of the wheatbelt region with extensive stands of living sheoak (Casuarina obesa) and paperbark (Melaleuca strobophylla) across the lake floor (occurrences other than Toolibin Lake)*. Department of Conservation and Land Management, Perth.

Heitmeyer M.E. (2010). *A Manual for Calculating Duck-Use-Days: To determine habitat resource values and waterfowl popultion energetic requiremets in the Mississippi Alluvial Valley*. Greenbrier Wetland Services, Bloomfield MO.

Jaensch R.P., Vervest R.M. & Hewish M.J. (1988). *Waterbirds in nature reserves of south-western Australia 1981-85: reserve accounts*. Royal Australasian Ornithologists Union, Melbourne.

Johnson R.K., Wiederholm T. & Rosenberg D.M. (1993). Freshwater Biomonitoring Using Individual Organisms, Populations and Species Assemblages of Benthic Macroinvertebrates. In: *Freshwater Biomonitoring and Benthic Macroinvertebrates*. (Eds D.M. Rosenberg & V.H. Resh), Chapman & Hall.

Lane J.A.K., Clarke A.G. & Winchcombe Y.C. (2017). *South west wetlands Monitoring Program 1977 - 2016*. Department of Biodiversity, Conservation and Attractions.

Legendre P. & Gallagher E.D. (2001). Ecologically meaningful transformations for ordination of species data. *Oecologia* **129**, 271–280

Leung Y.F.J. (2009). *Reproduction of the zooplankton, Daphnia carinata and Moina australiensis: implications as live food for aquaculture and utilization of nutrient loads in effluent (doctoral dissertation)*. The University of Adelaide, Adelaide, South Australia.

Maher M. & Carpenter S.M. (1984). Benthic studies of waterfowl breeding habitat in south-western New South Wales. II. Chironomid populations. *Australian Journal of Marine and Freshwater Research* **35**, 97–110

Martin J. (2013). *Australian Chironomus v. December 2013*. University of Melbourne. http://www.chironomidae.net/Martin/AustChironfiles/AustChironomusv1213.pdf; accessed April 2019.

McMahon G. (2006). *Ecological character description of Toolibin lake, Western Australia*. Department of Conservation and Land Management.

Muirden P.D. & Coleman S. (2014). *The Toolibin Natural Diversity Recovery Catchment: Review of Surfacewater Monitoring*. Department of Parks & Wildlife, Western Australia.

Munro D.R. (1975). *Lake Toolibin Report*. Department of Fisheries & Wildlife Western Australia.

Northern Arthur River Wetlands Committee (1987). *The Status and Future of Lake Toolibin as a Wildlife Reserve*. Water Authority, Western Australia.

Oksanen J., Blanchet F.G., Kindt R., Legendre P., Minchin P., O'Hara R.B., *et al.* (2013). *vegan: Community Ecology Package*.

Pinder A.M., Halse S.A., McRae J.M. & Shiel R.J. (2009). Aquatic invertebrate assemblages of wetlands and rivers in the wheatbelt region of Western Australia. Appendices. In: *Data appendices : a biodiversity survey of the Western Australian agricultural zone*. (Eds M.N. Lyons & Dunlop, J A), Department of Environment and Conservation.

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

Pinder A.M., Halse S.A., Shiel R.J. & McRae J.M. (2010). An arid zone awash with diversity: patterns in the distribution of aquatic invertebrates in the Pilbara region of Western Australia. *Records of the Western Australia Museum Supplement* **78**, 205–246

R Core Team (2014). *R: A language and environment for statistical computing.* R Foundation for Statistical Computing, Vienna, Austria.

Rabalais N.N., Diaz R.J., Levin L.A., Turner R.E., Gilbert D. & Zhang J. (2010). Dynamics and distribution of natural and human caused hypoxia. *Biogeosciences* **7**, 585–619

Rutherford J. (2015). *Hydrology Program Review 1977-2015: Toolibin Lake Natrural Diversity Recovery Catchment*. Department of Parks and Wildlife, Kensington, Western Australia.

Stokes R.A. & Sheridan R.J. (1985). *Hydrology of Lake Toolibin*. Water Authority of Western Ausrtralia, Leederville, Western Australia.

Svetlichny L.S. & Hubareva E.S. (2002). Effect of oxygen concentration on metabolism and locomotory activity of Moina micrura (Cladocera) cultured under hypo- and normoxia. *Marine Biology* **141**, 145–151. https://doi.org/10.1007/s00227-002-0805-x

Toolibin Lake Recovery Team and Toolibin Lake Technical Advisory Group (1994). *TOOLIBIN LAKE RECOVERY PLAN*. Toolibin Lake Recovery Team and Toolibin Lake Technical Advisory Group, Perth.

Zhang J., Gilbert D., Gooday A.J., Levin L., Naqvi S.W.A., Middelburg J.J., *et al.* (2010). Natural and human-induced hypoxia and consequences for coastal areas: synthesis and future development. *Biogeosciences* **7**, 1443–1467. https://doi.org/10.5194/bg-7-1443-2010

Appendices

Appendix 1. Invertebrate species by sample matrix, for Toolibin and Walbyring Lakes for April and October 2017. Invertebrates are recorded as estimated log abundance class per sample, i.e. 1-10 individuals =1, 11-100=2 etc.

		Toolibin						Walbyring					
		April			October			April			October		
LowestID	LowestIDNC	E	S	W	E	S	W	W1a	W2	W3	W1b	W2	W3
hydrazoan													
Hydra sp.	IB010199				2	2	2						
flatworm													
Turbellaria	IF999999												1
Microturbellaria	IF9999A0				1	1	1					1	1
round worm													
Nematoda	II999999								1	1			
mollusc													
Planorbidae	KG079999											1	
earth worms													
Dero digitata	LO050201				2	3	3	1		1	2	3	2
water mite													
Eylais sp.	MM030199					1							
Acercella falcipes	MM170101				1	1	1				1	1	
Pezidae	MM259999	1											

		Toolibin						Walbyring					
		April			October			April			October		
LowestID	LowestIDNC	E	S	W	E	S	W	W1a	W2	W3	W1b	W2	W3
Oribatida sp.	MM9999A1			1							1		1
Mesostigmata	MM9999A2	1		1									
cladoceran													
Leydigia australis	OG031802											3	
Pleuroxus inermis	OG032502				3	4	4				3	4	3
Daphnia carinata	OG040201	2	2	2									
Daphnia cephalata	OG040202				2	2	3				4	3	4
Simocephalus elizabethae	OG040505										3	3	3
Simocephalus gibbosus	OG040506				1	2							
Simocephalus victoriensis	OG040507										3	3	3
Moina sp.	OG0701A1	4	4	4				4	5	4			
ostrocod													
Alboa worooa	OH080101					2							
Bennelongia sp.	OH080399										3	3	3
Candonocypris novaezelandiae	OH080403											3	
Cypretta baylyi	OH080501										3	3	3
Heterocypris sp.	OH081099				1								3
Sarscypridopsis aculeata	OH090101				3	3	3				4	4	4
copepod													
Boeckella triarticulata	OJ110101				3	3	5				4	5	4

		Toolibin						Walbyring					
		April			October			April			October		
LowestID	LowestIDNC	E	S	W	E	S	W	W1a	W2	W3	W1b	W2	W3
Calamoecia sp. 342	OJ1102A1				2	2	3					4	3
Australocyclops australis	OJ310301	3	4	3	3	4	3	2	3	4	3	4	4
Mesocyclops brooksi	OJ310703				3		2			3		3	
Eucyclops australiensis	OJ311001					3	3						3
Paracyclops chiltoni	OJ3111A0				2								
amphipod-shrimp													
Austrochiltonia subtenuis	OP020102				1	1	1						
beetle													
Allodessus bistrigatus	QC091101							1					
Antiporus sp. (larvae)	QC091699				1		1				1	1	
Megaporus sp. (larvae)	QC092199					2							
Rhantus sp. (larvae)	QC092399	1	1		1	1		1			1	1	1
Lancetes lanceolatus	QC092401						1						
Eretes australis	QC092901	1								1			
Onychohydrus scutellaris	QC093401				1	1							
Bidessini (larvae)	QC0999A6		1			1							
Berosus munitipennis	QC110418		1										
Berosus nutans	QC110421		1										
Berosus sp. (larvae)	QC110499				1	1	1	1		1		1	
Enochrus elongatulus	QC111101	1						1					

		Toolibin						Walbyring					
		April			October			April			October		
LowestID	LowestIDNC	E	S	W	E	S	W	W1a	W2	W3	W1b	W2	W3
Enochrus sp. (larvae)	QC111199		1		1	1							
Paracymus sp.	QC111699		1										
Hydraena luridipennis	QC130101	1											
cranefly													
Tipulidae	QD019999						1						
mosquito													
Anopheles annulipes s.l.	QD070101					1				1			
Anopheles sp.	QD070199	1	1										
Culex (*Culex*) *australicus*	QD070706	2	2	1				2	1	1			
Culex (*Culex*) *annulirostris*	QD070709	1	1							2			
biting midge													
Atrichopogon sp. 2 (SAP)	QD0927A0								1	1			
Psychodinae sp. 2 (SAP)	QD1299A1				1								
soldier fly													
Stratiomyidae	QD249999		2		1	2		1	1	1	1	1	2
Ephydridae sp. 1 (SAP)	QD7899A5	1	1	1				1		1			
fly													
Muscidae sp. H (SAP)	QD8999A7	1											

		Toolibin						Walbyring					
		April			October			April			October		
LowestID	LowestIDNC	E	S	W	E	S	W	W1a	W2	W3	W1b	W2	W3
non-biting midge													
Procladius paludicola	QDAE0803		1		2	3	3				3	3	1
Procladius villosimanus	QDAE0804				1		1						
Ablabesmyia aff *notabilis*	QDAE11A2					3							
Paralimnophyes pullulus	QDAF1202						1					2	
Tanytarsus fuscithorax/semibarbitarsus	QDAH04D8				3	4	3					3	3
Chironomus occidentalis	QDAI0408	4	4	4				4	3	4			
Chironomus tepperi	QDAI0414				3	4	3				3	3	3
Chironomus aff. *alternans*	QDAI04A0	3	3	3	4	4	4	3	2	3	4	4	4
Dicrotendipes conjunctus	QDAI0603				3	3	3				3	3	3
Kiefferulus intertinctus	QDAI0701				3	3							
Kiefferulus martini	QDAI0703	1			3	3							
Polypedilum nubifer	QDAI0804				3	3	3				3	3	3
Cryptochironomus griseidorsum	QDAI1901						4					2	
waterstrider													
Microvelia (*Pacificovelia*) *oceanica*	QH560101	1	1	1		1				1			
boatmen													
Sigara truncatipala	QH650204	1	1										

		Toolibin						Walbyring					
		April			October			April			October		
LowestID	LowestIDNC	E	S	W	E	S	W	W1a	W2	W3	W1b	W2	W3
Agraptocorixa parvipunctata	QH650302				3	2	3				2	2	2
Agraptocorixa sp.	QH650399	1	1					1		1			
Micronecta robusta	QH650502		1		2	2	2				1	2	2
Micronecta gracilis	QH650503	1	1										
backswimmer													
Anisops thienemanni	QH670401				3	3	3				2	3	3
Anisops hyperion	QH670402						3						
Anisops gratus	QH670403	2	2	1	3								
Anisops sp.	QH670499							1	1	2			
damselfly													
Xanthagrion erythroneurum	QO021301				1	1							
Coenagrionidae	QO029999	1											
Austrolestes analis	QO050101				1	1	2					1	
Austrolestes annulosus	QO050102				1	1	1				1		
Austrolestes io	QO050105					1	1					1	
dragonfly													
Anax papuensis	QO121204	2	1	1	1	1	1	1	1	1	1	1	1
Hemicordulia tau	QO300102		1		1	2	1				1	2	1
caddisfly													
Oecetis sp.	QT250799						2						

		Toolibin						Walbyring					
		April			October			April			October		
LowestID	LowestIDNC	E	S	W	E	S	W	W1a	W2	W3	W1b	W2	W3
Triplectides australis	QT251103				1	2	3				1	1	1
Leptoceridae	QT259999	1											

Appendix 2 The relative abundance (%) of rotifer species in samples from Toolibin and Walbyring Lakes in April 2017.

Provided by Dr R. Shiel from a count of the first 200 individuals in the <250µm fraction of plankton samples (see methods).

		Toolibin			**Walbyring**		
Taxon	**Lowest Id**	**W**	**E**	**S**	**1**	**2**	**3**
Protista							
Ciliophora	*Euplotes* sp.				0.5	0.5	
Rhizopoda	*Arcella hemisphaerica*	0.5			0.5		0.5
Rotifera							
Bdelloidea	Indeterminate bdelloid	0.5	1.0	0.5		0.5	0.5
Asplanchnidae	*Asplanchna* brightwellii	0.5		0.5	4.6	5.9	3.6
Brachionidae	*Brachionus angularis* s.l.	0.5	2.0	0.5	1.3	15.3	46.9
	Brachionus budapestinensis			0.5	4.1	4.5	2.6
	Brachionus calyciflorus amphiceros			0.5	12.8	32.7	16.1
	Brachionus cf. *durgae*			0.5			
	Brachionus quadridentatus			0.5		0.5	
	Brachionus lyratus s.l.	0.5		0.5			
Dicranophoridae	*Dicranophoroides caudatus*					0.5	
Lepadellidae	*Lepadella* sp.	0.5			0.5		
Notommatidae	*Cephalodella tenuiseta*	0.5					
Synchaetidae	*Polyarthra vulgaris*	16.3					
	Polyarthra sp.			14.8	9.7	1.5	1.6
Trichocercidae	*Trichocerca similis*	0.5					
	Trichocerca sp.	0.5					
Trochosphaeridae	*Filinia* sp.	79.3	94.0	8.6	56.4	37.6	27.6
	indet. toed rotifer		1.0				