The Living Murray Condition Monitoring at Hattah Lakes 2015–16
Part A – Main Report

Prepared by: David Wood, Fiona Freestone, Paul Brown, Cherie Campbell and Scott Huntley

July 2016

Final Report

MDFRC Publication 118

The Living Murray Condition Monitoring at Hattah Lakes 2015–16
Part A – Main Report

Final Report prepared for the Mallee Catchment Management Agency by The Murray–Darling Freshwater Research Centre.

Mallee Catchment Management Authority
DEPI Complex
Cnr Koorlong Avenue & Eleventh Street
Irymple VIC 3498

Ph: (03) 5051 4377

This report was prepared by The Murray–Darling Freshwater Research Centre (MDFRC). The aim of the MDFRC is to provide the scientific knowledge necessary for the management and sustained utilisation of the Murray–Darling Basin water resources. The MDFRC is a joint venture between La Trobe University and CSIRO.

For further information contact:

Paul Brown

The Murray–Darling Freshwater Research Centre
PO Box 4095
Mildura VIC 3502
Ph: (03) 5051 4050

Email: Paul.Brown@latrobe.edu.au
Web: www.mdfrc.org.au
Enquiries: mdfrc@latrobe.edu.au

Report Citation: Wood D, Freestone F, Brown P, Campbell C, Huntley S (2016) The Living Murray Condition Monitoring at Hattah Lakes 2015–16 Part A – Main Report. Final Report prepared for the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre, MDFRC Publication 118/2016, July, 102 pp.

Cover Image: Moira grass (*Pseudoraphis spinescens*) in Chalka Creek.

Photographer: Fiona Freestone

Document history and status

Version	Date issued	Reviewed by	Approved by	Revision type
Draft v1	6 June 2016	Paul Brown	David Wood	Project leader review
Draft v2	9 June 2016	Kerry Whitworth	Paul Brown	Content and copy edit
Draft v3	30 June 2016	Andrew Greenfield, MCMA; MDBA	Paul Brown	Client review

Distribution of copies

Version	Quantity	Issued to
Final v4	1 word version & 1 PDF	Andrew Greenfield, MCMA

Filename and path:	\\SHE - Life Sciences\MDFRC\Projects\Mallee CMA\TLM CM\648 TLM CM 2015-16\01 Hattah\Reporting\Draft Report
Author(s):	David Wood, Fiona Freestone, Paul Brown, Cherie Campbell and Scott Huntley
Author affiliation(s):	The Murray–Darling Freshwater Research Centre
Project Manager:	Paul Brown
Client:	Mallee Catchment Management Authority
Project Title:	TLM Condition Monitoring – Hattah Lakes Icon Site
Document Version:	Draft
Project Number:	M/BUS/648
Contract Number:	15.1387

Finalised 29 July 2016

Acknowledgements:

This project was undertaken with the support of the Murray–Darling Basin Authority; the Victorian Department of Environment, Land, Water and Planning; and Parks Victoria. The assistance provided by Jenny Collins, Louise Chapman, Andrew Greenfield, Peter Kelly and Nicholas Sheahan from the Mallee Catchment Management Authority is greatly appreciated.

The authors would also like to thank present and former MDFRC staff for their input into project design and development, including: Braeden Lampard, Taylar Pay, Greg Cranston, Danielle Linklater and Deborah Bogenhuber for their field and laboratory assistance; Helen Missen and La Trobe staff for their administrative assistance and Kerry Whitworth for editing this report.

This project was funded by The Living Murray initiative of the Murray–Darling Basin Authority. The Living Murray is a joint initiative funded by the New South Wales, Victorian, South Australian, Australian Capital Territory and Commonwealth governments, coordinated by the Murray–Darling Basin Authority.

The Murray–Darling Freshwater Research Centre offices are located on the land of the Latje Latje and Wiradjuri peoples. We undertake work throughout the Murray–Darling Basin and acknowledge the traditional owners of this land and water. We pay respect to Elders past, present and future.

Contents

List of figures

List of tables

Executive summary

This report details the condition monitoring undertaken at the Hattah Lakes icon site as part of The Living Murray (TLM) Condition Monitoring Program. Icon site condition monitoring has been developed to:

- determine the change in environmental condition of individual assets resulting from water application and the implementation of works programs under The Living Murray scheme
- assess whether sustainable native fish and vegetation communities are being maintained across the icon sites.

This report documents inter-annual change in the whole-of-icon-site condition with respect to floodplain trees (River Red Gum and Black Box), vegetation communities (wetland and floodplain), Lignum and fish assemblages. In a broad sense this involves assessing progress towards the achievement of ecological objectives specified in the Environmental Water Management Plan 2012.

River Red Gum

As of 2015–16, it is deemed that the overall health of River Red Gum is being maintained above current targets, therefore the specific adopted objective *Sustainable populations of River Red Gum* is being met for River Red Gum communities at the Hattah Lakes icon site.

River Red Gum tree condition is generally good at Hattah Lakes icon site and is being maintained above target values for 2015–16. Germination of large numbers of seedlings has occurred on multiple occasions since 2010. These events have coincided with natural flooding (2010–11) and environmental watering (2013 and 2014) and are particularly common in strand-lines around waterbodies. Seedling density at one location was estimated to be as high as 2000 seedlings m^{-2}, while seedlings thought to be two years old (2013 germination) were found at densities of 1000 seedlings m^{-2}.

In 2015–16 the trend of net population growth (which has been occurring since at least 2012–13) has continued, with recruitment into the River Red Gum population higher than mortality.

Black Box

As of 2015–16, it is deemed that the overall health of Black Box is being maintained above current targets, therefore the specific adopted objective *Sustainable populations of Black Box* is being met for Black Box communities at the Hattah Lakes icon site.

Black Box condition was above the target value in 2015–16, as it has been since at least 2010–11. This improvement is likely a result of increased rainfall and flooding since the end of the millennium drought.

Seedling germination was more predominant in Black Box swampy woodland than in riverine chenopod woodland. This may mirror the inundation extents of recent flooding and environmental watering. The riverine chenopod woodland is located higher on the floodplain and so was not inundated to the same extent as the lower Black Box swampy woodland. In 2015–16, annual recruitment was keeping pace with mortality.

Wetland vegetation communities

Favourable environmental conditions have benefited the wetland water-responsive plant community at Hattah Lakes. The lakes were inundated by floodwater in 2010–11 and environmental water has been delivered over three consecutive years since then (2013–15). The majority of wetlands were inundated at the time of the 2015–16 surveys.

A total of 74 plant species was recorded, with the most abundant being Red Water-milfoil (*Myriophyllum verrucosum*), an amphibious floating species. The seven most abundant plant species recorded in 2015–16 were water-responsive. Three species were recorded for the first time in nine years of TLM condition monitoring at Hattah Lakes wetlands. Four of the species recorded are listed as having conservation significance in Victoria. A diversity of habitat types has been supported within the Hattah Lakes wetlands in 2015–16 (e.g. inundation to different depths, recent drawdown and damp mud, and flow recession in the last year or two). This habitat mosaic supports species diversity and abundance.

Species richness was in line with expectations for inundated wetlands. It is anticipated that the composition and abundance of amphibious species will increase in future surveys as wetlands continue to draw down. Comparisons of ongoing condition monitoring data (as wetlands draw down) to vegetation responses following natural floods (e.g. 2011–12 and 2012–13 compared to 2015–16) are recommended and will be useful in guiding the delivery of future environmental flows.

Floodplain vegetation communities

Sections of the water-responsive plant community on the Hattah Lakes floodplain have benefited from a return to favourable environmental conditions since 2010. This is likely in response to flows inundating a portion of the floodplain on multiple occasions over the last few years (e.g. flooding in 2010–11 followed by delivery of environmental water in 2013, 2014 and 2015). While water-responsive plant species in River Red Gum communities have benefited from repeated inundation, most Black Box understorey vegetation communities have not been inundated for an extended period of time (~20 years).

Overall, species richness in 2015–16 was the second-highest over nine years of condition monitoring. This is likely a result of water recession from the floodplain following repeat inundation. A total of 113 species were recorded across all floodplain sites at Hattah Lakes, including eight species listed as having conservation significance in Victoria. Ten species were recorded for the first time at Hattah floodplain sites in the last nine years of TLM condition monitoring. Two of these species are listed as vulnerable in Victoria: Jerry-jerry (*Ammannia multiflora*) and Glistening Dock (*Rumex crystallinus*).

Consecutive environmental flows appear to be supporting flow-dependent rare plants on the Hattah Lakes floodplain. There is limited information about these species, largely because of their ephemeral nature. Targeted surveys (timed to coincide with the drawdown of wetlands following the delivery of environmental water) are recommended and would provide valuable information for the management of flow-dependent rare plants.

Lignum

Over 10 years of TLM condition monitoring of Lignum, Hattah Lakes has experienced severe drought (mid-1990s to 2009), followed by widespread flooding (late 2010–11). Areas of floodplain that were not inundated by overbank flooding received substantial rainfall during the summer of 2010–11. A portion of the Lignum community was inundated by environmental flows in 2014 and 2015.

Overall, Lignum at Hattah Lakes was in moderate condition in 2015–16. The condition of Lignum has declined since 2012–13 as the floodplain continues to dry following natural flooding, indicating that the effects of the flood (e.g. soil moisture on the floodplain) are diminishing across the icon site. Lignum condition in 2015–16 more closely resembles that recorded in 2009–10, which is contradictory to expectations. Half of the sites surveyed had received environmental water during 2014 and so it was anticipated that there would be an improvement in condition between the

2013–14 and 2015–16 surveys. It is possible that condition has deteriorated at sites that did not receive environmental water, counteracting the benefits of water delivery at an icon-site scale. An analysis of the differences between long-dry sites and sites that received environmental water is recommended to improve our understanding of Lignum condition and response to environmental watering at Hattah Lakes.

Fish

Relatively high abundances of local wetland fish in the Hattah Lakes indicate ongoing recruitment since the previous monitoring period (2013–14). Therefore, the objective *Increase distribution, number and recruitment of local wetland fish* is currently being met. The second objective for fish at Hattah Lakes, *Maximise use of floodplain habitat for recruitment of all indigenous freshwater fish*, is being met for most small-bodied fish in Hattah Lakes, with ongoing presence indicating recruitment. Large-bodied fish recruitment has occurred but, as it included recruitment via connectivity with the Murray River (immigration and emigration), it is difficult to determine precisely using the current monitoring plan. To better monitor large-bodied fish recruitment, the current monitoring plan needs to be complemented by intervention monitoring programs.

In 2015–16, six wetlands (two were dry) within the Hattah Lakes icon site, as well as the adjacent Murray River, were surveyed for fish. Species diversity was similar to other survey years. Relatively high abundances of many small-bodied native fish species were caught within the wetlands compared to previous surveys. However, the abundance of the non-native Eastern mosquitofish was also the highest of all sampling years. There was a general lack of large-bodied fish species (native and non-native) caught within the wetlands, with many expected native species (known to be present through sampling in intervention monitoring programs) not found during the current sampling program.

The Murray River adjacent to Hattah Lakes supports a healthy Murray cod population, with the highest abundance of fish of all sampling years caught in 2015–16. Of this, 20% of the catch was comprised of young-of-year Murray cod, indicating a recent successful breeding season. The abundance of Golden perch in the Murray River in 2015–16 was also the highest of all sampling years.

1 Introduction

1.1 Purpose of the report

This report details the condition monitoring undertaken at Hattah Lakes as part of The Living Murray Condition Monitoring Program 2006–07 to 2015–16, which was funded by the Murray–Darling Basin Authority. This work was conducted by The Murray–Darling Freshwater Research Centre (MDFRC) for the Mallee Catchment Management Authority (MCMA). The MDFRC is a research centre of La Trobe University. This report represents a deliverable requirement for Contract No. 15.1397 between La Trobe University and the MCMA.

1.2 Report structure

The Hattah Lakes Condition Monitoring Report for 2015–16 consists of two parts: Part A comprises the main report and Part B the supplementary material. Part A contains structured reports for each of the ecological components monitored (River Red Gum, Black Box, wetland vegetation, floodplain vegetation, Lignum, and fish). Part B contains material that supports Part A content (e.g. site information, species lists, photo plates, etc.).

The objectives and mode of delivery for TLM condition monitoring at the Hattah Lakes have, to this reporting period, remained largely consistent for the duration of the program. Condition monitoring was unfunded in the previous survey period, 2014–15. Therefore, this report presents results of monitoring during 2015–16, after a two-year gap for most ecological components. The only sampling that took place during this 'gap' was for tree structure and this has been included where possible.

A review of the monitoring and reporting framework commenced in July 2013 (Robinson 2013). This review process was ongoing during 2014–15 (Brown et al. 2015a; Robinson 2014a, b) and identifies a number of means by which monitoring and reporting may be improved. While the present report incorporates recommendations of the review as a means of progressing refinement of the reporting process, the authors use a combination of previously employed reporting mechanisms as well as a number of new indices, as specified in the previous condition monitoring report (Henderson et al. 2014).

1.3 The Hattah Lakes

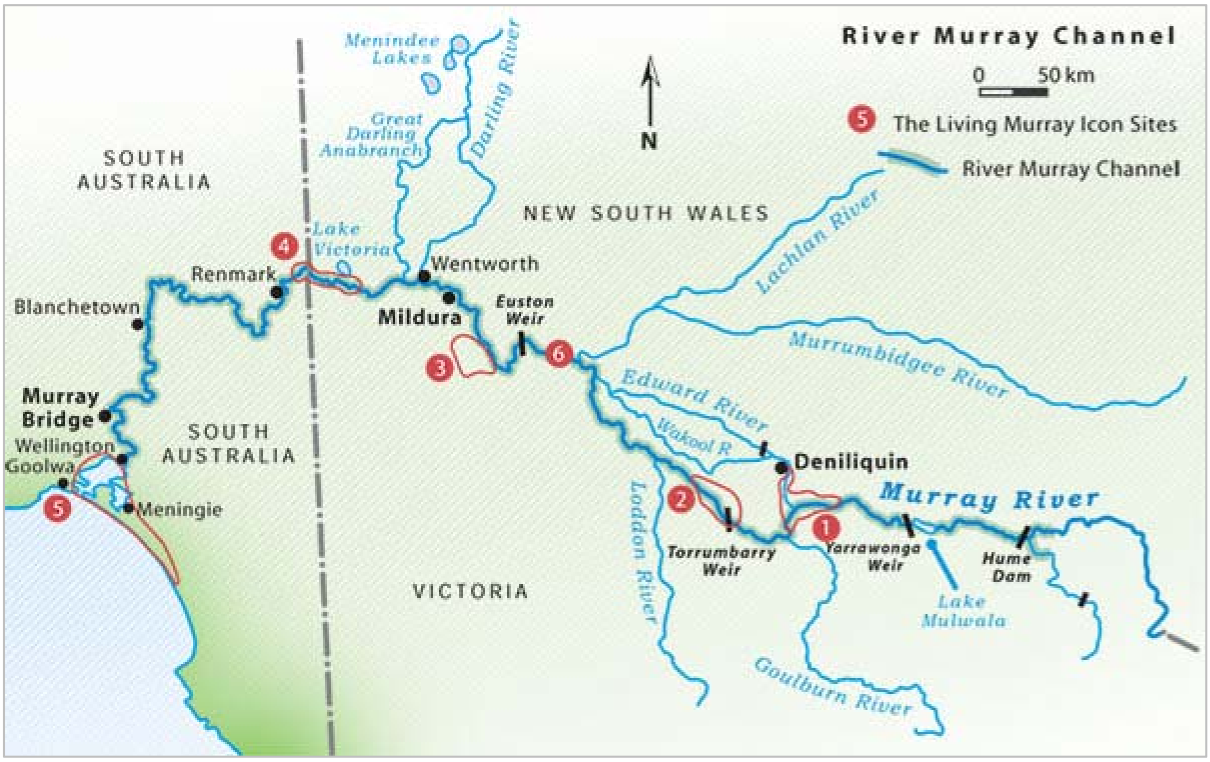


Figure 1. Locations of TLM icon sites: (1) Barmah–Millewa Forest, (2) Gunbower–Koondrook–Perricoota Forest, (3) the Hattah Lakes, (4) Chowilla Floodplain and Lindsay-Wallpolla Islands, (5) the Lower Lakes, Coorong and the Murray Mouth and (6) the Murray River Channel (image courtesy of MDBA).

The Hattah Lakes icon site is one of six sites identified as ecologically significant under TLM's first step (MDBMC 2003) (Figure 1). The icon site is part of the 48 000 ha Hattah–Kulkyne National Park located in the north-west of Victoria (Figure 2). The Hattah Lakes system contains 18 freshwater lakes, 12 of which are Ramsar-listed (Butcher & Hale 2011), connected by a series of floodplain channels fed by the Murray River during periods of high flow. The mosaic of water bodies includes creeks and lakes of varying depths and acts as a sink, or store, for nutrients and sedimentary deposits including plant and animal propagules from the surrounding catchments (MDBC 2006). The Hattah Lakes have significance in protecting endangered species of flora and fauna in Australia and provides important refuges for a range of biota including fish, birds and vegetation. Hattah Lakes also has significant social and cultural value, having provided sanctuary for Indigenous society for thousands of years. The economic values of Hattah Lakes include recreational and tourism values as well as the provision of flood control and a potential emergency water supply for the local township of Hattah (MDBC 2006). More detailed information on the economic, social, cultural and environmental values of the Hattah Lakes icon site is contained in The Living Murray Foundation Report (MDBC 2005) and in the Environmental Management Plan (MDBC 2006).

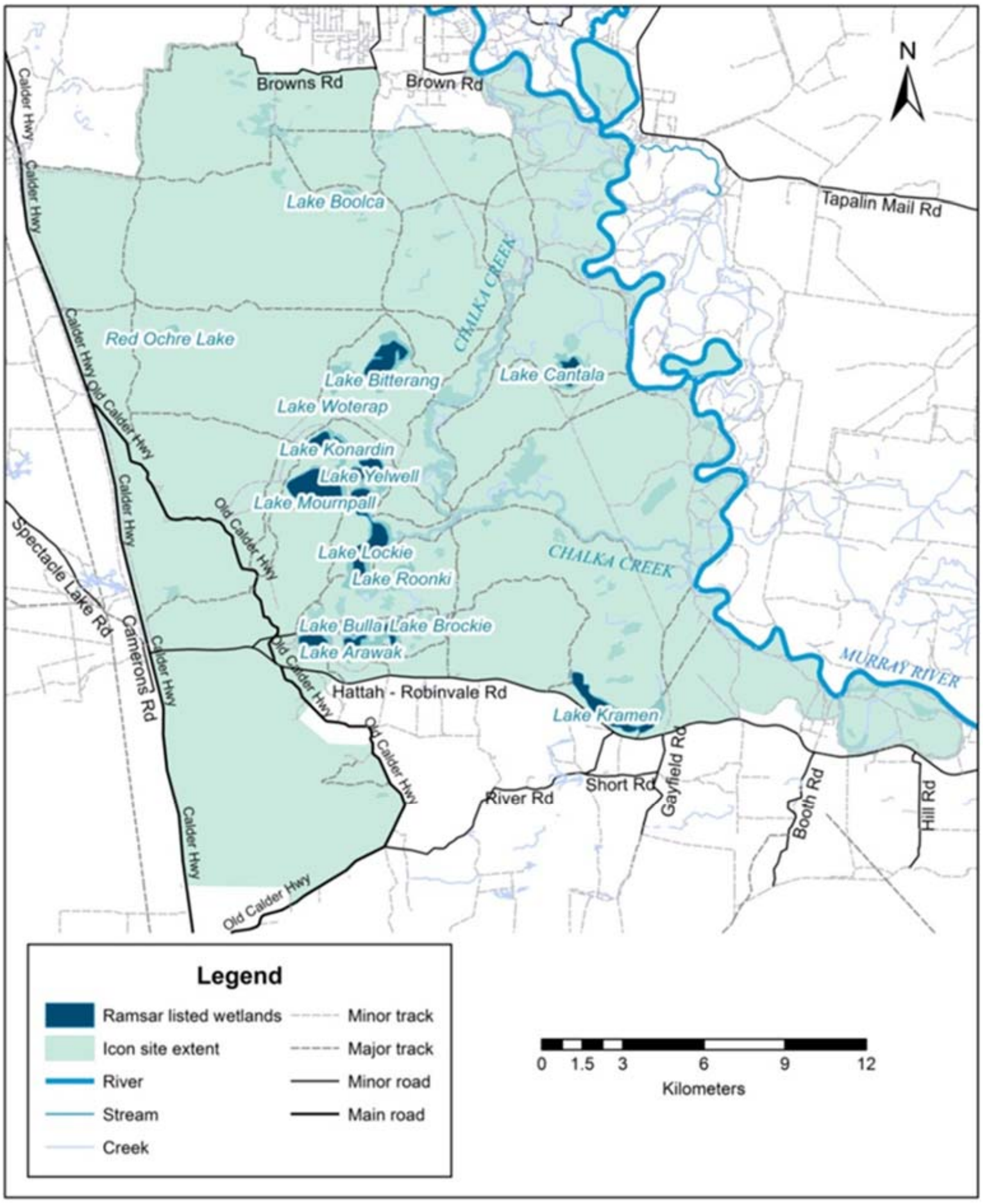


Figure 2. Hattah Lakes icon site (Ramsar-listed lakes shown in dark blue).

1.3.1 Hydrology

A major risk to the floodplain communities at the Hattah Lakes is altered water regimes (Butcher & Hale 2011; MDBA 2012a; MDBC 2006). There have been substantial reductions in the timing, frequency, duration and magnitude of flooding at Hattah Lakes as a result of river regulation and water extraction (Ecological Associates 2007; MDBC 2006; SKM 2003). Extraction of water from the Murray River upstream of Hattah Lakes has resulted in the mean discharge in the Murray River near Hattah Lakes being approximately 50% of the natural value (Maheshwari et al. 1993). The timing of flooding is delayed from the natural by approximately two months (shifted from August to October). Flooding frequency is reduced by 57% and duration by 65% (MDBC 2006).

To overcome these changes in hydrology, pumping has become necessary to maintain the ecological condition of the lakes. Between 2005 and 2010, transportable pumps were used to deliver water into the Hattah Lakes from the Murray River. In late 2009 and 2010 heavy rains caused localised flooding and this was followed in late-2010 to mid-2011 by overbank flooding (Figure 3). In October 2013, a permanent pump station, regulators and stop banks were built on the floodplain. This new infrastructure was used to deliver 61 GL of water via Chalka Creek to the lakes and surrounding low-lying floodplain between October 2013 and January 2014. The partial recession of water throughout 2014 was followed by a further 'top-up' flow, delivering 92 GL via the pump station during winter 2014. The lakes and wetlands filled again and surrounding, more elevated floodplain was inundated. In the spring of 2014 regulators on north and south Chalka Creek were opened to allow a controlled discharge of environmental water to return from the floodplain to the Murray River (Brown et al. 2015b). This is the first time that environmental watering via pumps and regulators has been used to simulate the two-way connection between the Murray River and its floodplain wetlands at such a scale during an otherwise low-flow period (Figure 3).

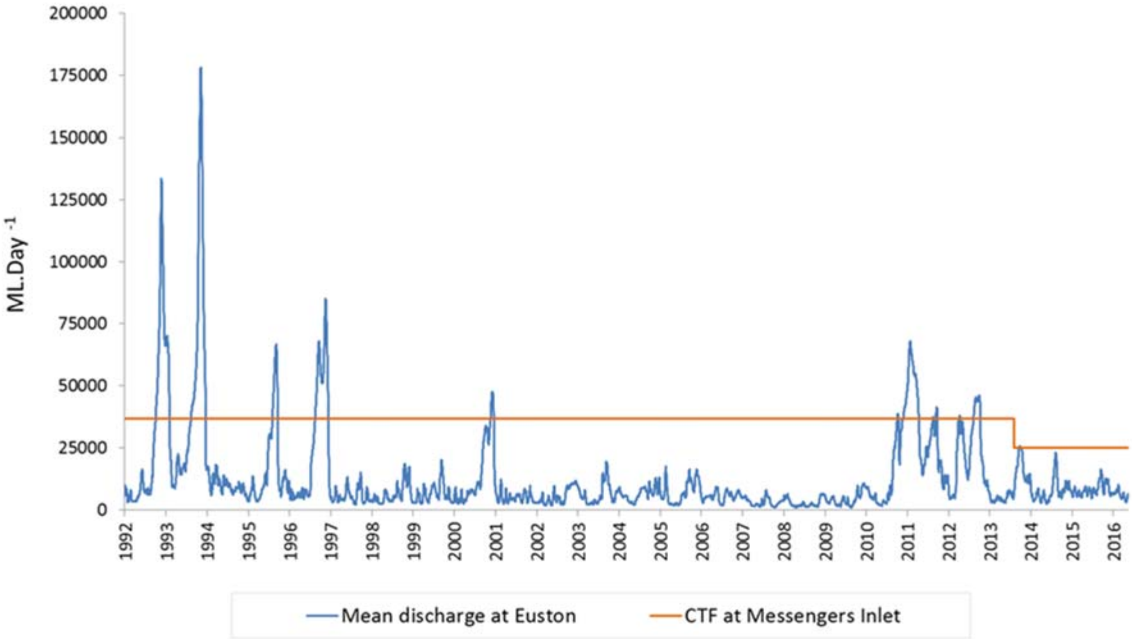


Figure 3. Historical discharge (ML.day^{-1}) at Euston weir on the Murray River 1992 to June 2016 is shown in blue. Commence to flow (CTF) for Chalka Creek (36 700 ML.day^{-1} prior to October 2013, approximately 25 000 ML.day^{-1} thereafter) is shown in orange.

2 River Red Gum

DAVID WOOD

2.1 Introduction

River Red Gum (*Eucalyptus camaldulensis*) is widespread throughout the Murray–Darling Basin. In the Hattah–Kulkyne National Park, River Red Gum communities are most common along the Murray River and adjacent flood runners and wetlands. Flooding is an integral part of the River Red Gum's ecology and provides an important source of water to maintain populations. Changes in flooding regime and groundwater status threaten the condition, recruitment and long-term sustainability of River Red Gums, particularly on the lower Murray River floodplain (Maheshwari et al. 1995; MDBC 2006). The Living Murray program aims to maintain the condition and extent of River Red Gum communities at Hattah Lakes through environmental works and the delivery of environmental water.

Condition monitoring reports on the change in environmental condition at the icon-site scale. Monitoring is specifically tailored to determine if management objectives are being met. River Red Gum is monitored on an annual basis as outlined in the Condition Monitoring Program design for Hattah Lakes (MDFRC 2011).

2.2 Ecological objectives

Ecological objectives for the Hattah Lakes have been under refinement since interim objectives were first developed by the Murray–Darling Ministerial Council in 2003 (MDBMC 2003). The most recent version of the ecological objective for River Red Gum is based on an understanding of environmental responses learned through monitoring, evaluation, research, and modelling and consultation activities over nine years (MDBC 2006). The ecological objective for River Red Gum is:

> *Maintain and, where practical, restore the ecological character of the Ramsar site with respect to the Strategic Management Plan.*

The specific adopted objective resulting from the refinement process (Robinson 2014a) is:

> *Sustainable populations of River Red Gum.*

2.3 Methods

Two methods were employed to assess the condition of River Red Gum at Hattah Lakes: (i) tree condition monitoring and (ii) population demographics. To allow for assessment and comparison at the River Red Gum community scale, sites were stratified within River Red Gum communities. These are differentiated as water regime classes (WRC; Table 2.1).

Comprehensive details on tree condition monitoring and size-class distribution methods are available in the Condition Monitoring Program design for the Hattah Lakes (MDFRC 2011).

Table 2.1. The water regime classes and their component ecological vegetation classes (EVCs) used to define River Red Gum communities at Hattah Lakes based on hydrological association (Ecological Associates 2007).

Water regime class	Area (ha)	Component EVCs	Characteristics
Red Gum forest	341	106 Riverine grassy forest 811 Grassy riverine forest	Found only in areas subject to the most frequent flooding regimes. This water regime class is subject to inundation in nearly all years, and winter floods maintain the density, size and health of River Red Gum.
Fringing Red Gum woodland	2739	813 Intermittent swampy woodland 818 Shrubby riverine woodland	Occurs mainly in floodplain areas immediately surrounding wetlands and along water courses that are inundated by peaks in river flow during most years.
Red Gum with flood-tolerant understorey	1533	295 Riverine grassy woodland	Represents the driest habitat for River Red Gum. Floods in this WRC are intermittent and brief.

2.3.1 Tree condition

Tree condition monitoring is a ground-based monitoring method used to detect changes in River Red Gum condition based on assessing a number of variables for each tree (MDBA 2012b). For each sample tree crown extent, crown density, new tip growth, epicormic growth, leaf die-off, bark cracking, reproductive extent and mistletoe load were scored and the diameter-at-breast-height (DBH) measured (MDBA 2012b).

Twenty-seven sites, each comprising of 30 River Red Gum trees, were established in 2007–08 and sampled annually to 2015–16 (with the exception of five sites in 2010–11, which could not be accessed due to flooding, and all sites in 2014–15 when the program was unfunded).

To compensate for loss of sample trees due to mortality, for each live tree lost a replacement was randomly selected (next closest live tree). Accordingly, only the live tree component of the sample set for any given year is considered when comparing inter-year differences in tree condition. Dead tree data were used to calculate mortality rates but not used in assessing changes in condition between years. For more detailed information on site establishment, locations and sampling refer to MDFRC (2011).

2.3.2 Population demographics

Population size-class distribution surveys are used to inform population status assessments and, in conjunction with tree condition monitoring (mortality data), to inform population growth assessments. These assessments are used to evaluate long-term sustainability of River Red Gum at Hattah Lakes and relate closely to the objective of restoring healthy floodplain communities (MDBA 2012c).

Size-class distribution of River Red Gum is assessed on a three-year rolling cycle such that for each year approximately one third of sites are sampled. Transects were established in 2006–07, 2007–08 and 2008–09, covering 52.8 ha, which represents approximately 1.14% of the extent of River Red Gum at Hattah Lakes.

Each transect was navigated end-to-end using a hand-held GPS. Each River Red Gum tree within the transect had its DBH measured and its position recorded. While DBH may not be a consistent indicator of age for an individual tree (Roberts & Marston 2011; Snowball 2001) in the absence of a suitable alternative it is used here as a proxy where it is assumed that, on average, the larger the DBH of the tree the older it is.

Data from sites first surveyed in 2006–07 and reassessed in 2009–10, 2012–13 and 2015–16 are presented in this report. To examine temporal trends in population structure, all live trees were classified into 15-cm DBH categories. Counts were square-root transformed to adjust for the high proportion of seedlings.

2.4 Indices and points of reference

The identification of suitable indices and associated points of reference for reporting on the condition and maintenance of River Red Gum are currently being developed as part of a program design refinement process (Robinson 2013). As part of this process a revised reporting framework is being developed for implementation in 2016–17. In the interim, this report uses a combination of both previously used reporting mechanisms and more recently recommended measures to evaluate and report River Red Gum condition.

2.4.1 Tree condition

The target developed for River Red Gum at Hattah is:

- 85% of trees with crown extent score ≥ 4.

A crown extent score of equal to or greater than four is associated with a tree crown that is more than 40% foliated (Table 2.2). This point of reference is based on TLM condition monitoring data collected from 2007–08 to 2012–13, which indicate that River Red Gum trees with less than 40% foliated crown are at significantly higher risk of mortality than those with more foliated crowns (unpublished data).

Table 2.2. Category scale for reporting crown extent assessments (MDBA 2012b).

Score	Description	Percentage of assessable crown
0	None	0%
1	Minimal	1–10%
2	Sparse	11–20%
3	Sparse–Medium	21–40%
4	Medium	41–60%
5	Medium–Major	61–80%
6	Major	81–90%
7	Maximum	91–100%

The percentage of sampled trees with a crown extent score ≥ 4 was calculated per site and averaged across each WRC. The mean is the estimate of the frequency of trees within the population with a crown extent score ≥ 4 (Table 2.2). The standard error of the mean is expressed in plots as error bars (± SE).

2.4.2 Population demographics

Two methods for evaluating population status were used.

- population status index
- population growth index.

Population status

The *population status index* is based on the 'inverse J-shaped' curve (George et al. 2005), which is an ideal structure in sustainable tree populations. The method for calculation of the index is based on the example provided in Robinson (2013). The index was calculated as the difference (distance) between the rank order of the reference curve (i.e. inverse J-shaped curve) and the rank order of the sampled population for each site. This was then averaged for each WRC. The metric of comparison used was Spearman's Rho (ρ) which was then converted to an index value of between zero and one, as follows:

$$\rho = \frac{\sum_i (x_i - \bar{x})\,(y_i - \bar{y})}{\sqrt{\sum_i (x_i - \bar{x})^2 \sum_i (y_i - \bar{y})^2}}$$

$$Index = (\rho + 1)/2$$

Where X_i-X_n is a dataset of the ranked order of the reference curve, Y_i-Y_n is a dataset of the ranked order of the sample population, and $\bar{X}$ and $\bar{Y}$ are the sample means of those datasets, respectively.

Population growth

The *population growth index* was the net population growth for Red Gum forest, calculated as the difference between the three-year averages for recruitment and mortality. The approach for calculating recruitment was based on the current understanding of growth rates and estimated age at maturity and was calculated on population size-class data. River Red Gum growth rate data show that, on average, the DBH of River Red Gum trees at Hattah Lakes increases at a rate of approximately 1 cm per year (MDFRC, unpublished data). On the assumption that trees mature at approximately 10 years of age (George 2004), it follows that annual recruitment into the adult population may be measured as the number of trees that, when monitored once each year, record a DBH between 10 and 11 cm. To apply this approach to a population that is monitored on a three-year rolling basis (1/3 of sites monitored each year; each site revisited once every three years), annual recruitment into the adult population was calculated as one third of the number of trees with a DBH between 10 and 13 cm.

The annual mortality rate was calculated as the proportion of trees ≥ 10 cm DBH that died between one year and the next. Mortality rate calculations were performed on data collected as part of tree condition monitoring where individually tagged trees provided accurate records for tree deaths. For consistency, annual mortality rates were calculated as one third of the mortality rate associated with the preceding three-year period.

Net gain in adult trees and was derived by subtracting the average annual mortality rate from the average annual recruitment rate. A negative population growth index score therefore indicates a reduction in the growth of the adult population, and a positive score is indicative of an increase in adult tree population.

2.5 Results

2.5.1 Tree condition

For 2015–16, the target of 85% of River Red Gum trees exceeding a crown extent score of ≥ 4 was met for all water regime classes (Red Gum forest, Red Gum woodland and Red Gum with flood-tolerant understorey) at Hattah Lakes.

During 2008–09, River Red Gum at Hattah Lakes did not meet the target of a mean frequency of 85% trees with a crown extent value of ≥ 4 (Figure 4, Figure 5, Figure 6). In the survey year following 2008–09, mean frequency had increased for all water regime classes. However, it was not until 2010–11 that the target value was exceeded for all water regime classes. Since 2010–11 mean frequency has remained relatively stable above the target for all water regime classes. Sample photo point images in section 1 of Part B of this report give a visual reference of tree condition since 2011.

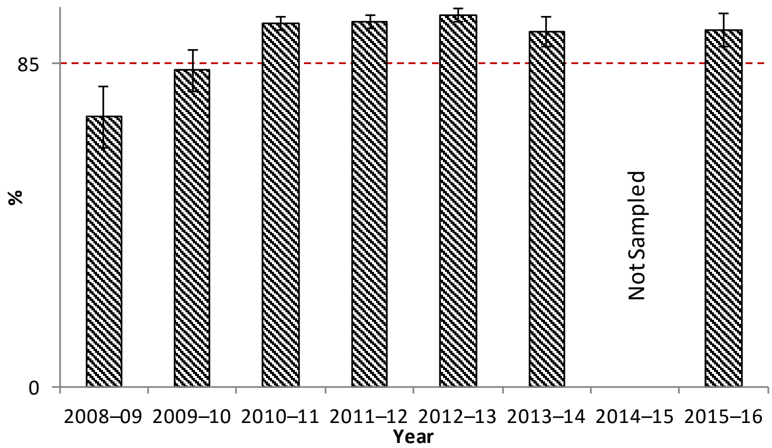


Figure 4. Mean frequency (± SE) of River Red Gum trees with crown extent scores ≥ 4 recorded in Red Gum forest at sites sampled annually in summer between 2008–09 and 2015–16 (except for 2014–15 when the program did not run).

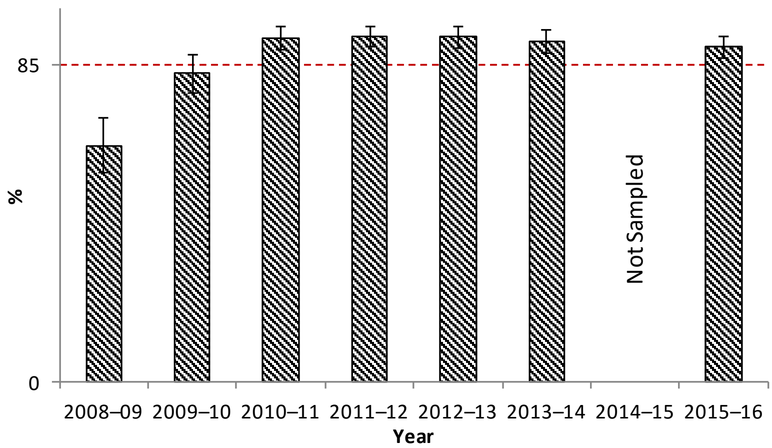


Figure 5. Mean frequency (± SE) of River Red Gum trees with crown extent scores ≥ 4 recorded in Red Gum woodland at sites sampled annually in summer between 2008–09 and 2015–16 (except for 2014–15 when the program did not run).

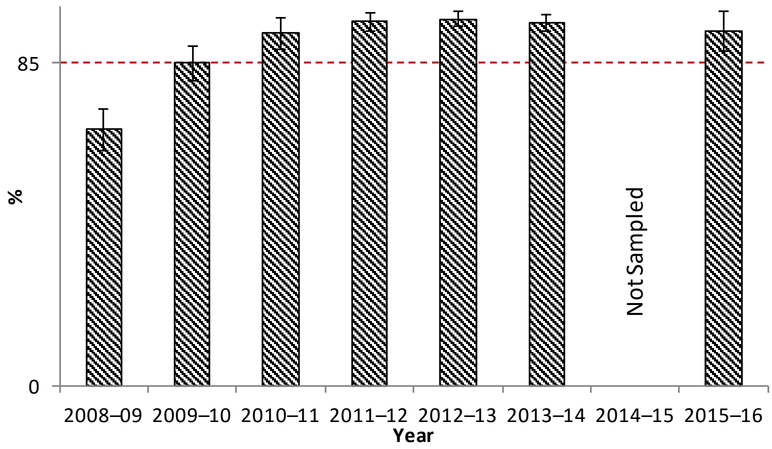


Figure 6. Mean frequency (± SE) of River Red Gum trees with crown extent scores ≥ 4 recorded in Red Gum with flood-tolerant understorey at sites sampled annually in summer between 2008–09 and 2015–16 (except for 2014–15 when the program did not run).

2.5.2 Population demographics

Population status

Size-class frequency distributions for all WRCs were similar for the periods 2006–07 and 2009–10, before a large increase in the number of 0–15-cm DBH trees occurred to 2012–13. Following this trend, a further increase in the 0–15-cm DBH size class, most apparent in Red Gum woodland and Red Gum with flood-tolerant understorey, occurred in 2015–16 (Figure 7, Figure 10 and Figure 13). In both instances recently germinated seedlings accounted for this increase (Figure 8, Figure 11 and Figure 14).

Population index status scores for River Red Gum forest show no change between monitoring periods, suggesting that the overall population structure has not changed substantially since 2006–07 (Figure 9).

An increase in the mean Red Gum woodland population index status score has occurred since 2009–10 (Figure 12) indicating a population becoming more closely aligned with the ideal ‘inverse J-shaped’ curve, which is indicative of a sustainable population.

Population structure for River Red Gum with a flood-tolerant understorey has shown marginal change between periods (Figure 15). However, there is no clear directional trend in changes since 2006–07.

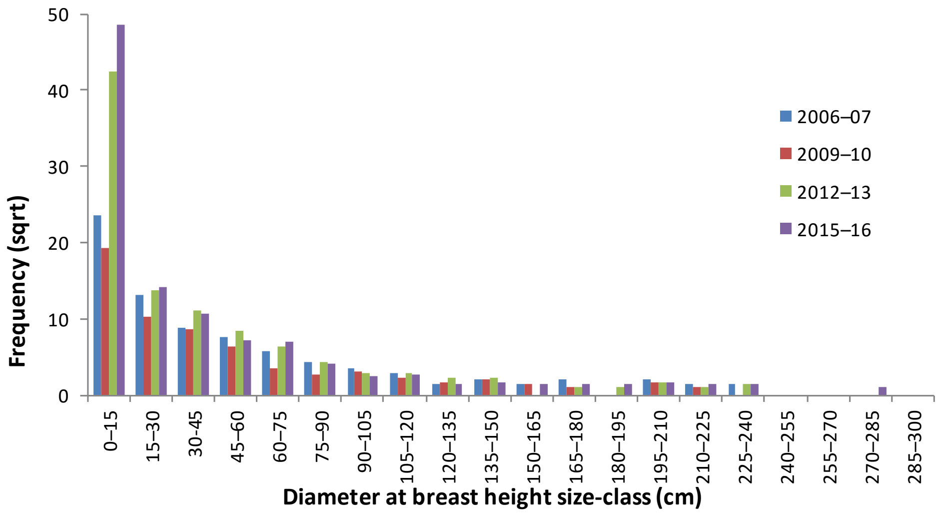


Figure 7. Size-class distribution of live River Red Gum forest trees (0–300 cm DBH) at Hattah Lakes; n(2006–07) = 962, n(2009–10) = 641, n(2012–13) = 2286, n(2015–16) = 2830.

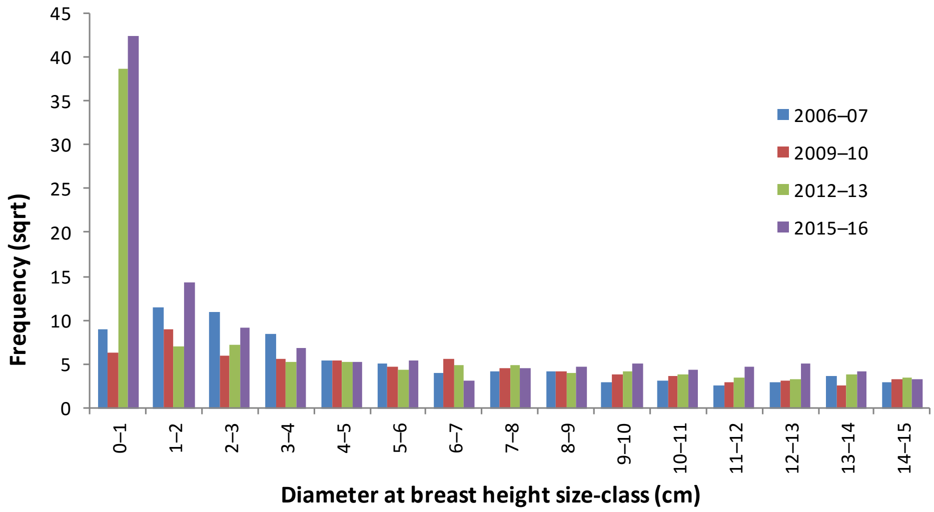


Figure 8. Size-class distribution of live River Red Gum forest trees (0–15 cm DBH) at Hattah Lakes; n(2006–07) = 560, n(2009–10) = 375, n(2012–13) = 1809, n(2015–16) = 2363.

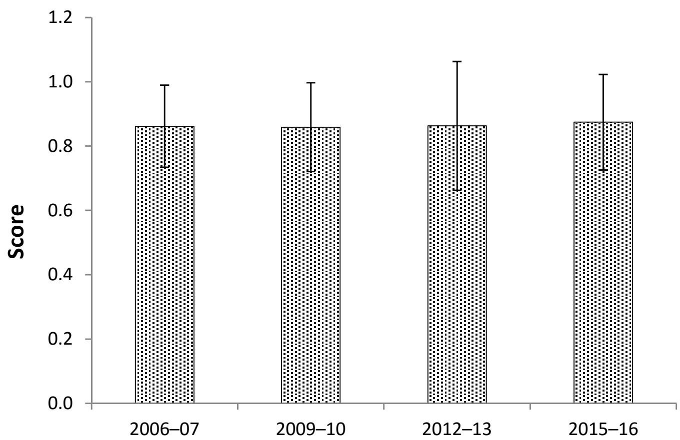


Figure 9. Population status index (± 95% CI) for Red Gum forest calculated based on level of correlation with the reference 'inverse J-shaped' curve.

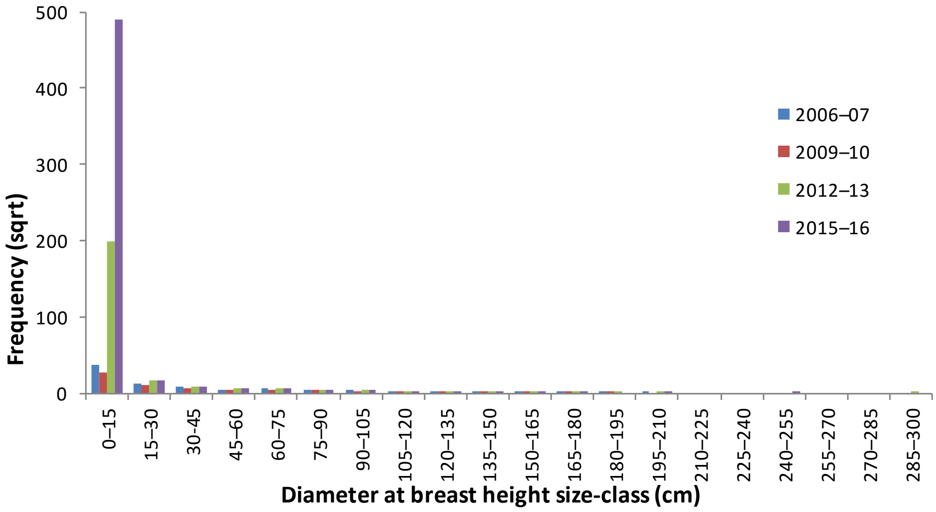


Figure 10. Size-class distribution of live River Red Gum woodland trees (0–300 cm DBH) at Hattah Lakes; n(2006–07) = 1674, n(2009–10) = 960, n(2012–13) = 39 892, n(2015–16) = 240 682.

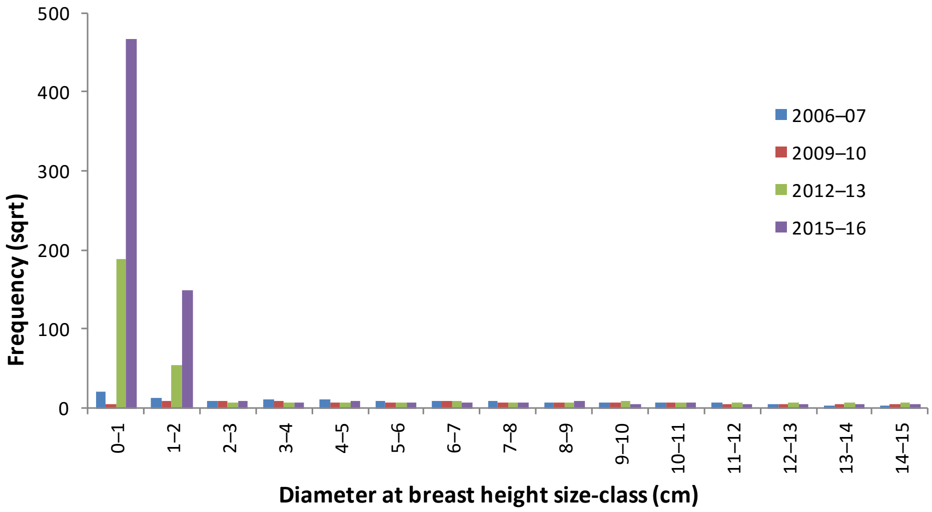


Figure 11. Size-class distribution of live River Red Gum woodland trees (0–15 cm DBH) at Hattah Lakes; n(2006–07) = 1320, n(2009–10) = 696, n(2012–13) = 39 414, n(2015–16) = 240 196.

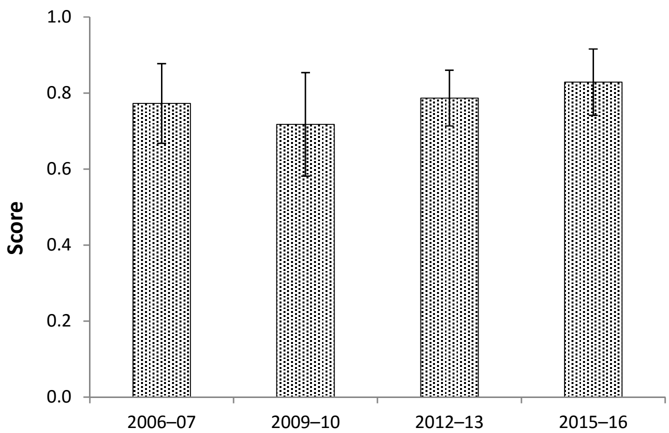


Figure 12. Population status index (± 95% CI) for Red Gum woodland calculated based on level of correlation with the reference 'inverse J-shaped' curve.

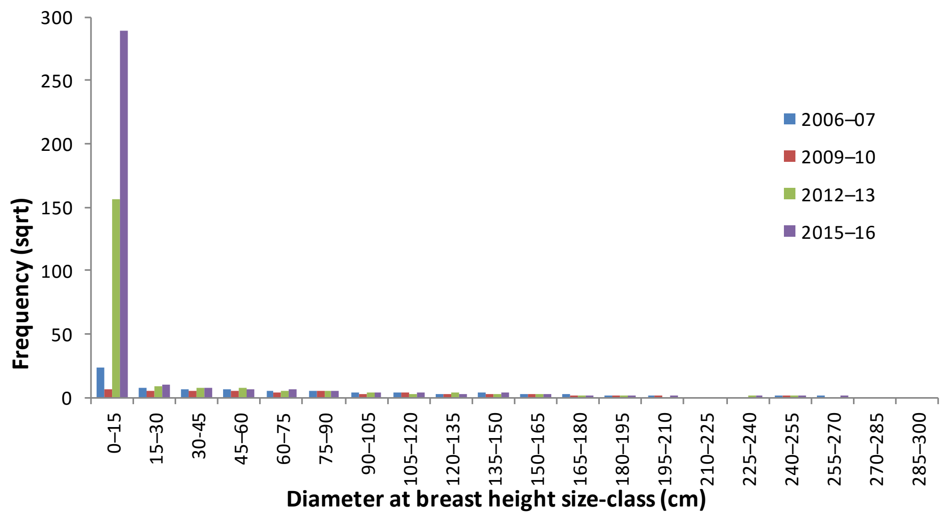


Figure 13. Size-class distribution of live River Red Gum with flood-tolerant understorey trees (0–300 cm DBH) at Hattah Lakes; n(2006–07) = 811, n(2009–10) = 222, n(2012–13) = 24 825, n(2015–16) = 83 869.

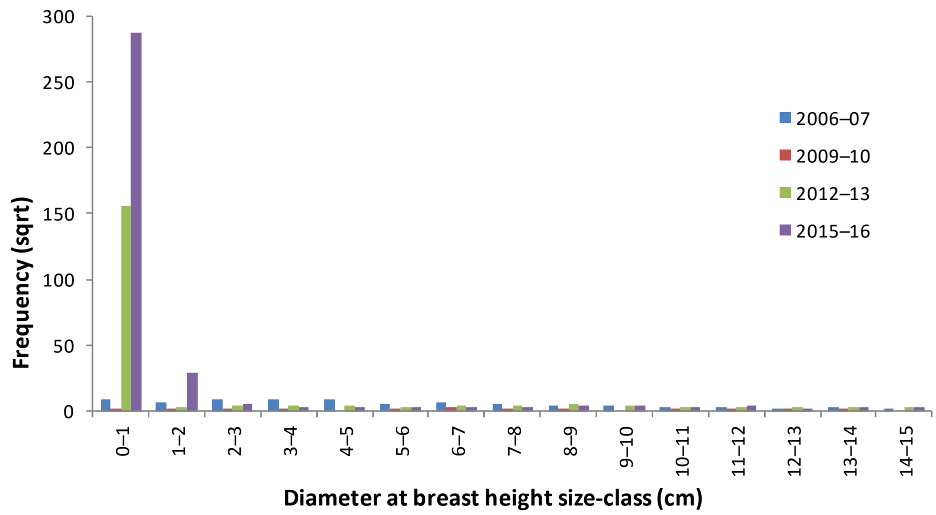


Figure 14. Size-class distribution of live River Red Gum with flood-tolerant understorey trees (0–15 cm DBH) at Hattah Lakes n(2006–07) = 532, n(2009–10) = 46, n(2012–13) = 24 530, n(2015–16) = 83 568.

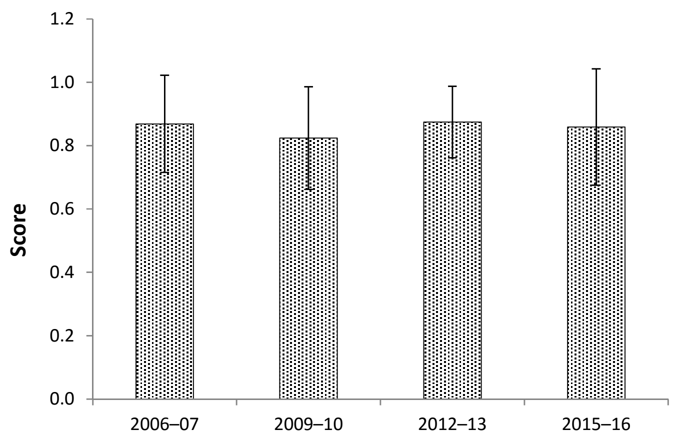


Figure 15. Population status index (± 95% CI) for Red Gum with flood-tolerant understorey calculated based on level of correlation with the reference 'inverse J-shaped' curve.

Population growth

Net population growth occurred in 2015–16 for all three River Red Gum WRCs at Hattah Lakes. For the Red Gum forest, population growth has occurred annually since 2011–12 (Figure 16), while for the Red Gum woodland and Red Gum with flood-tolerant understorey classes, population growth has occurred annually since 2012–13 (Figure 17 and Figure 18). Annual mortality prior to 2012–13 was greater across all three WRCs than post 2012–13 (with the exception of 2008–09).

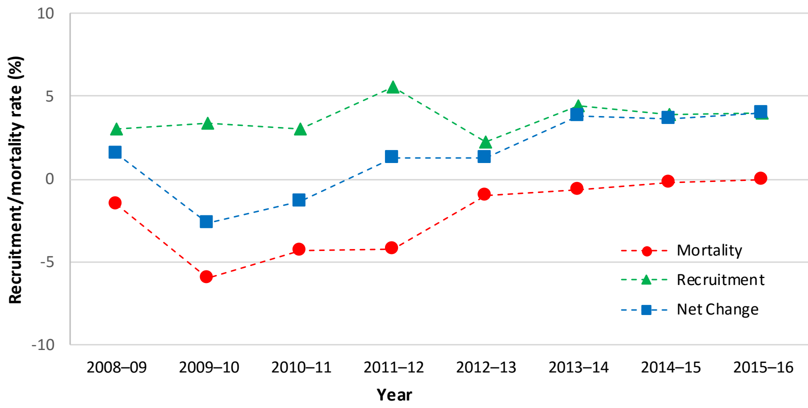


Figure 16. Net annual population growth for River Red Gum in Red Gum forest, calculated as the difference between recruitment and mortality.

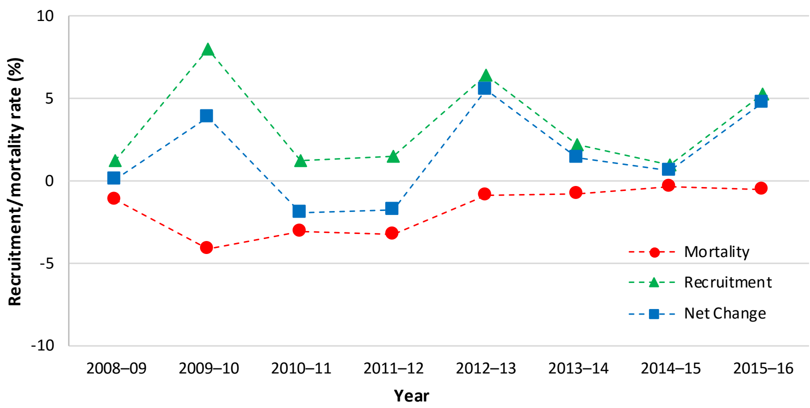


Figure 17. Net annual population growth for River Red Gum in Red Gum woodland, calculated as the difference between recruitment and mortality.

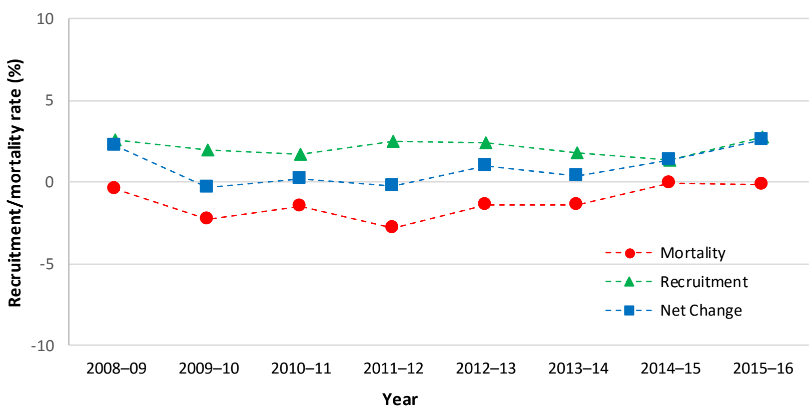


Figure 18. Net annual population growth for River Red Gum in Red Gum with flood-tolerant understorey, calculated as the difference between recruitment and mortality.

2.6 Discussion

While each of the River Red Gum water regime classes experience slightly different environmental conditions, particularly with reference to flood frequency, they have all shown similar changes in condition with regards to crown extent score. Flooding of Hattah Lakes (natural or artificial) over the past five years has influenced all three water regime classes, not just the lowest on the floodplain (Red Gum forest) which is inundated with small-scale environmental watering.

Environmental watering of Hattah Lakes was first undertaken in 2006 with the specific aim of restoring and improving the health of River Red Gums fringing the lakes. Below-average rainfall continued until 2010. While the environmental watering between 2006 and early 2010 helped to improve the condition of River Red Gums surrounding the Hattah Lakes (McCarthy et al. 2009), the area of influence was relatively localised. The condition of other River Red Gums at Hattah Lakes remained relatively poor.

During 2010 and 2011 above-average rainfall and natural flooding occurred. It was not until this time that wide-scale improvement in River Red Gum tree health was detected. Flooding in 2010–11 influenced the majority of the River Red Gum sites at Hattah Lakes, with high rainfall adding to available water sources. Groundwater in the region may have been recharged by excess water.

In the years following 2010–11, the region once again experienced below-average rainfall. A large area of Hattah Lakes was allocated environmental water during 2013 and 2014. This water was possibly sufficient to help many River Red Gums to improve or maintain condition. However, trees outside the area of influence of the environmental water (such as on Murray River bends) may have declined in condition during this period.

The River Red Gum population at Hattah Lakes has seen significant germination of seedlings since 2010–11. The initial germination was instigated by natural flooding in 2010–11 which inundated all three River Red Gum water regime classes. Since then, operation of the new pump and regulators at Hattah Lakes has occurred with environmental watering in 2013 and then again in 2014, to a greater extent. The 2014 watering inundated a large proportion of floodplain (5583 hectares at 45 m AHD), much of which had not been inundated for approximately 20 years.

Following each addition and recession of water over these two years, large numbers of River Red Gum seedlings germinated. Seedling germination was particularly common in strand-lines around waterbodies. At some locations newly germinated seedling density was estimated at as high as 2000 seedlings m^{-2}. Seedlings thought to be two years old (2013 germination) were found at densities of 1000 seedlings m^{-2}. For comparison, peak seedling density at Hattah Lakes is magnitudes greater than for River Red Gum on the Ord River in Western Australia (27 seedlings m^{-2}) (Pettit & Froend 2001).

The highest rates of germination in 2015–16 were recorded in the Red Gum woodland and Red Gum with flood-tolerant understorey water regime classes. These two water regime classes received environmental water in 2013 and 2014. Red Gum forest was not influenced to the same extent, as the majority of the area is located in the bends of the Murray River where environmental water did not reach.

While the vast majority of the seedlings that germinated since 2010–11 will perish over time, environmental conditions with a strong focus on flooding (i.e. frequency, depth and duration) will play an important role in seedling establishment and survival of these seedlings to mature trees.

While there appears to be large changes in the size-class distribution between years for both black box communities, the general shape of the distribution does not change (dominated by trees < 15 cm DBH). As the general shape of the distributions do not change substantially, there is little change in the population status index. This index is best for determining long-term trends in the populations of black box (and river red gum) trees as large changes to the overall population structure generally occur slowly (with the exception of considerable ecological disturbances).

Population growth has occurred annually since 2011–12 for the Red Gum forest and since 2012–13 for the Red Gum woodland and Red Gum with flood-tolerant understorey classes. This improvement is likely a result of the positive effects of natural flooding (2010–11) and two large-scale environmental waterings (2013 and 2014), whereby increased water availability has reduced mortality and increased growth. Prior to this, higher mortality was observed and, while environmental watering had been undertaken at Hattah Lakes, the extent did not influence as large an area as during later flooding.

As of 2015–16, it is deemed that the overall health of River Red Gum is being maintained above current targets. Therefore the specific adopted objective *Sustainable populations of River Red Gum* is being met for River Red Gum communities at the Hattah Lakes icon site.

3 Black Box

DAVID WOOD

3.1 Introduction

Black Box (*Eucalyptus largiflorens*) is a common flora species at Hattah Lakes and one of only a few large tree species in the area. Black Box generally occur higher on the floodplain (i.e. in less frequently flooded landscapes) than River Red Gum, although there is considerable overlap in their distributions. Both eucalypts play an ecologically similar role in their provision of carbon and habitat for floodplain flora and fauna (Briggs & Maher 1983; Mac Nally et al. 2001).

Black Box is a drought-tolerant, flood-responsive species that can adapt to varying environmental conditions through its ability to utilise water from floods, rainfall, creeks and groundwater (Holland et al. 2006; Jolly et al. 1993; McCarthy et al. 2009). As part of The Living Murray program, Black Box trees are monitored to ensure sustainable communities are maintained.

The Living Murray program aims to restore healthy floodplain communities at Hattah Lakes, of which Black Box comprise a significant proportion, through environmental works and the delivery of environmental water. Although management options to influence Black Box at Hattah Lakes have increased recently due to the 2013 works program, much of the Black Box community is still unable to be reached by environmental water.

Black Box is monitored on an annual basis as outlined in the Condition Monitoring Program design for Hattah Lakes (MDFRC 2011). Condition monitoring reports on Black Box condition and population status at the icon site scale. Monitoring is specifically tailored to determine if the ecological objective for Black Box is being met.

3.2 Ecological objectives

Ecological objectives for the Hattah Lakes have been in refinement since interim objectives were first developed by the Murray–Darling Basin Ministerial Council in 2003 (MDBMC 2003). The most recent version of the ecological objective for Black Box is based on an understanding of environmental responses learned through monitoring, evaluation, research, and modelling and consultation activities over nine years (MDBC 2006). The ecological objective for Black Box is:

> *Maintain and, where practical, restore the ecological character of the Ramsar site with respect to the Strategic Management Plan.*

The specific adopted objective resulting from the refinement process (Robinson 2014a) is:

> *Sustainable populations of Black Box.*

3.3 Methods

In order to address objectives relating to Black Box, two methods were employed to assess the condition of Black Box at Hattah Lakes: (i) tree condition monitoring and (ii) population demographics. To allow for assessment and comparison at the Black Box community scale, sites are stratified within Black Box communities. These are differentiated as WRCs (Table 3.1).

Comprehensive details on tree condition monitoring and size-class distribution assessments are provided in the Condition Monitoring Program design for the Hattah Lakes (MDFRC 2011).

Table 3.1. The water regime classes used to define Black Box communities at Hattah Lakes based on hydrological association (Ecological Associates 2007).

Water regime class	Area (ha)	Component EVCs	Characteristics
Black Box swampy woodland	6073	823 Lignum swampy woodland	Woodland commonly associated with Lignum understorey. Generally occurs at lower terraces and thus more prone to flooding than riverine chenopod woodland.
Riverine chenopod woodland	339	103 Riverine chenopod woodland (*syn*. Black Box chenopod woodland)	Woodland on most elevated riverine terraces. Naturally subject to only extremely infrequent shallow flooding from major events if flooded at all.

3.3.1 Tree condition

Tree condition monitoring is a ground-based monitoring method used to detect changes in Black Box condition based on assessing a number of variables for each tree (MDBA 2012b). For each sample tree crown extent, crown density, new tip growth, epicormic growth, leaf die-off, bark cracking, reproductive extent and mistletoe load were scored and the DBH measured (MDBA 2012b).

Eighteen sites, each comprising 30 Black Box trees, were established in 2007–08 and sampled annually to 2015–16; except for some sites that could not be reached in 2010–11 due to flooding and all sites in 2014–15 when the program was unfunded.

To compensate for loss of sample trees due to mortality, for each live tree lost a replacement was randomly selected (next closest live tree). Accordingly, only the live tree component of the sample set for any given year is considered when comparing inter-year differences in tree condition. For more detailed information on site establishment, locations and sampling refer to MDFRC (2011).

3.3.2 Population demographics

Size-class distribution surveys are used to inform population status assessments and, in conjunction with tree mortality data collected as part of tree condition monitoring, to inform population growth assessments. These assessments are used to evaluate the long-term sustainability of Black Box at Hattah Lakes and relate closely to the objective to sustain species assemblages typical of Black Box woodland (MDBA 2012c).

Size-class distribution of Black Box is assessed on a three-year rolling cycle such that for each year approximately one third of sites are sampled. Transects were established in 2006–07, 2007–08 and 2008–09, covering 25.1 ha, which represents approximately 0.39% of the areal extent of Black Box at the Hattah Lakes.

Each transect was navigated end-to-end using a hand-held GPS. Each Black Box tree within the transect had its DBH measured and its position recorded. As for Red Gum, DBH is used as a proxy for age.

Data from sites first surveyed in 2006–07 and reassessed in 2009–10, 2012–13 and 2015–16 are presented in this report. To examine temporal trends in population structure, all live trees were grouped into 15-cm DBH categories. Counts were square-root transformed to adjust for the high proportion of seedlings.

3.4 Indices and points of reference

As for River Red Gum, the identification of suitable indices and associated points of reference for reporting on the condition and maintenance of Black Box is currently under refinement (Robinson 2013). This report uses a combination of both previously used reporting mechanisms and more recently recommended measures to evaluate and report Black Box condition.

3.4.1 Tree condition

The target developed for Black Box tree condition at the Hattah Lakes icon site is:

- 80% of trees with crown extent score ≥ 4.

A crown extent score of equal to or greater than four is associated with a tree crown that is greater than 40% foliated (Table 2.2). This point of reference is based on TLM condition monitoring data collected from 2007–08 to 2012–13 that indicates Black Box trees with less than 40% foliated crown are at significantly higher risk of mortality (unpublished data).

The percentage of sampled trees with a crown extent score ≥ 4 was calculated per site and averaged across each WRC. The mean is the estimate of the frequency of trees within the population with a crown extent score ≥ 4 (Table 2.2). The standard error of the mean is expressed in plots as error bars (± SE).

3.4.2 Population demographics

Two methods for evaluating population status were used.

- population status index
- population growth index.

Population status

The population status index is based on the 'inverse J-shaped' curve (George et al. 2005) which is an ideal structure in sustainable tree populations. The index was calculated as described in section 2.4.2

Population growth

The population growth index was calculated as the net population growth for Black Box calculated as the difference between the cumulative three-year averages for recruitment and mortality. Annual recruitment and mortality rates and the net gain in adult trees were calculated as described in section 2.4.2. As for River Red Gum, Black Box growth rate data show that, on average, trees at Hattah Lakes grow at a rate of approximately 1 cm per year (MDFRC, unpublished data). It was assumed that Black Box trees mature at approximately 10 years of age (George 2004).

For the Black Box swampy woodland water regime class no sites were assessed for population growth during some years (second year of the three-year rolling cycle; 2010–11 and 2013–14). Lack of representative sites in some years results in an incomplete time series and as such has not been reported upon.

3.5 Results

3.5.1 Tree condition

For 2015–16, the target of 80% of Black Box trees exceeding a crown extent score of ≥ 4 was met for both WRCs (Black Box swampy woodland and riverine chenopod woodland) at Hattah Lakes.

The mean frequency (± SE) of the crown extent score improved markedly between 2008–09 and 2010–11 for both the Black Box swampy woodland and riverine chenopod woodland WRCs (Figure 19 and Figure 20). Since 2010–11 there has been little variation in the mean frequency (± SE) for either water regime class. Trees in Black Box swampy woodland have maintained a crown extent score of ≥ 4 in greater than 80% of the population since 2009–10. Trees in the river chenopod woodland WRC have maintained a crown extent of ≥ 4 in greater than 80% of the population since 2010–11. Sample photo point images in section 2 of Part B of this report give a visual reference of tree condition since 2011.

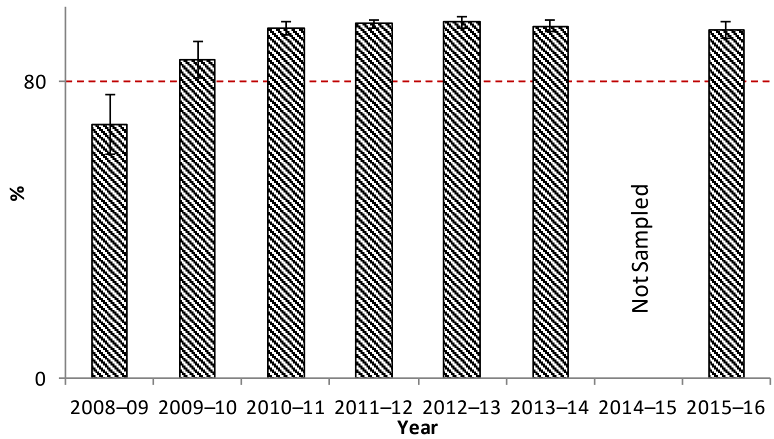


Figure 19. Mean frequency (± SE) of Black Box trees with crown extent scores ≥ 4 recorded in Black Box swampy woodland at sites sampled annually in summer between 2008–09 and 2015–16 (except for 2014–15 when the program did not run).

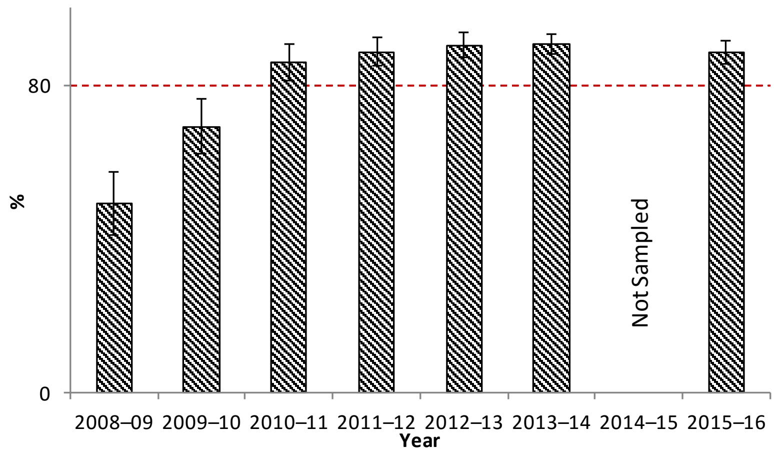


Figure 20. Mean frequency (± SE) of Black Box trees with crown extent scores ≥4 recorded in riverine chenopod woodland at sites sampled annually in summer between 2008–09 and 2015–16 (except for 2014–15 when the program did not run).

3.5.2 Population demographics

Population status

Size-class frequency distributions for Black Box swampy woodland were similar for 2006–07 and 2009–10. During 2012–13 there was a marked increase in 0–15-cm DBH trees, which was repeated in 2015–16 (Figure 21). Since 2012–13 further germination has occurred, which was evident in 2015–16 as an increase in the number of trees less than 1 cm DBH (Figure 22). While the mean population status of Black Box swampy woodland improved slightly during 2012–13, a decline occurred to 2015–16 (Figure 23). This indicates a gradual shift away from what is considered a sustainable population structure. A substantial degree of variability also exists within the Black Box swampy woodland population, as highlighted by large error bars.

Size-class frequency distributions for riverine chenopod woodland show little variability between sampling periods, with the exception of 2009–10, when a reduced number of trees were sampled across all size-classes (Figure 24). Additionally, during 2006–07 a higher number of 0–15-cm DBH trees were evident than in other sampling periods. A shift in the size-class of newly germinated seedlings in 2012–13 to larger individuals (2–4 cm DBH) during 2015–16 also occurred (Figure 25; Figure 24). Overall, the population status index for riverine chenopod woodland Black Box trees remains relatively unchanged since 2006–07, indicating a stable population at the present time (Figure 26).

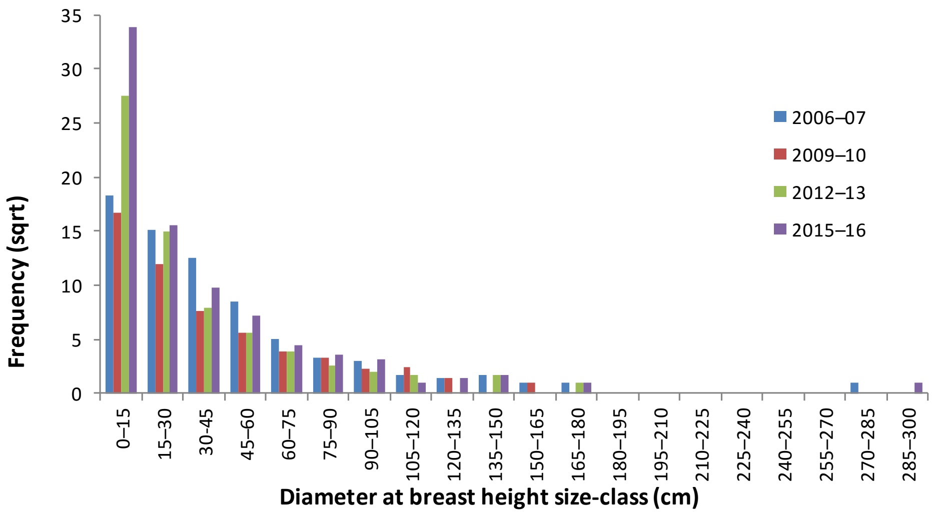


Figure 21. Size-class distribution of live Black Box swampy woodland trees (0–300 cm DBH) at Hattah Lakes; n(2006–07) = 846, n(2009–10) = 551, n(2012–13) = 1108, n(2015–16) = 1588.

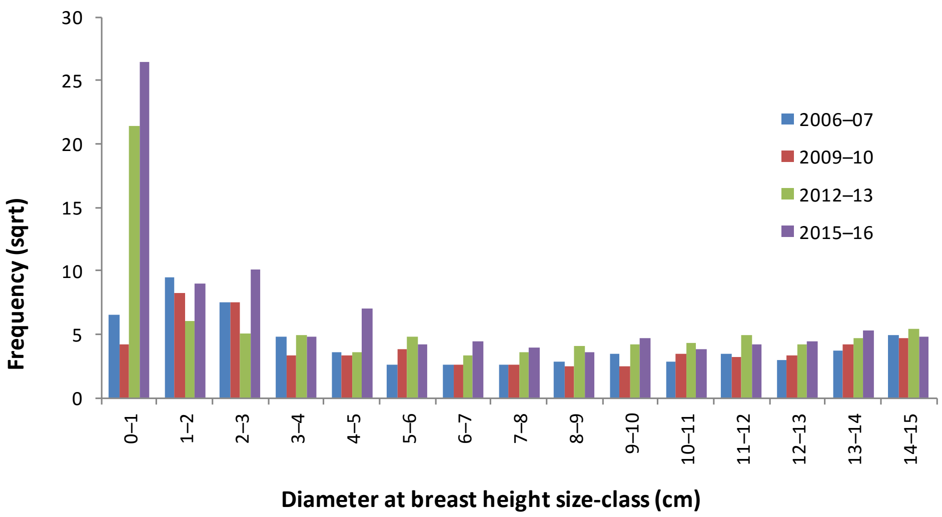


Figure 22. Size-class distribution of live Black Box swampy woodland trees (0–15 cm DBH) at Hattah Lakes; n(2006–07) = 333, n(2009–10) = 278, n(2012–13) = 757, n(2015–16) = 1147.

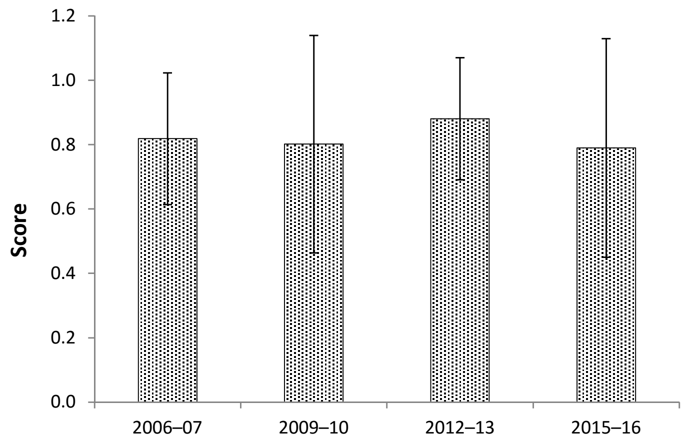


Figure 23. Population status index (± 95% CI) for Black Box swampy woodland calculated based on level of correlation with the reference 'inverse j-shaped' curve.

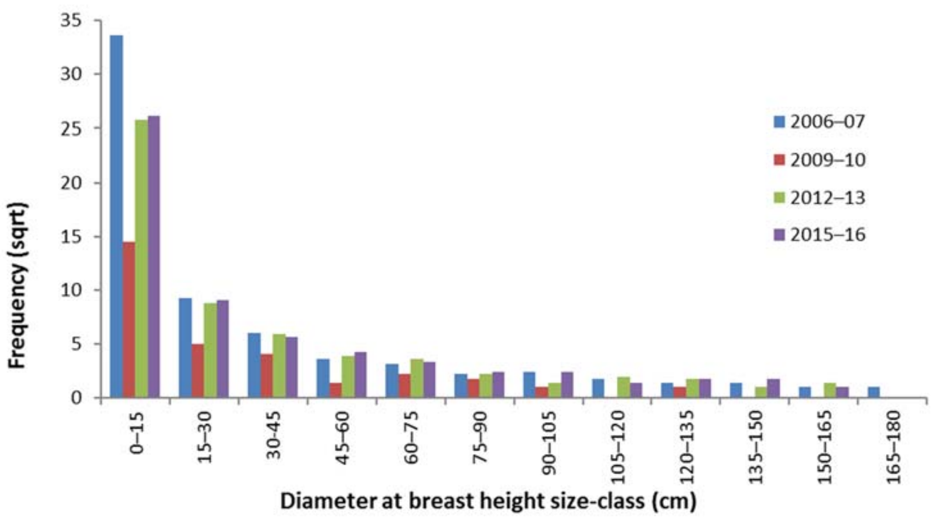


Figure 24. Size-class distribution of live riverine chenopod woodland trees (0–180 cm DBH) at Hattah Lakes; n(2006–07) = 1295, n(2009–10) = 264, n(2012–13) = 824, n(2015–16) = 845.

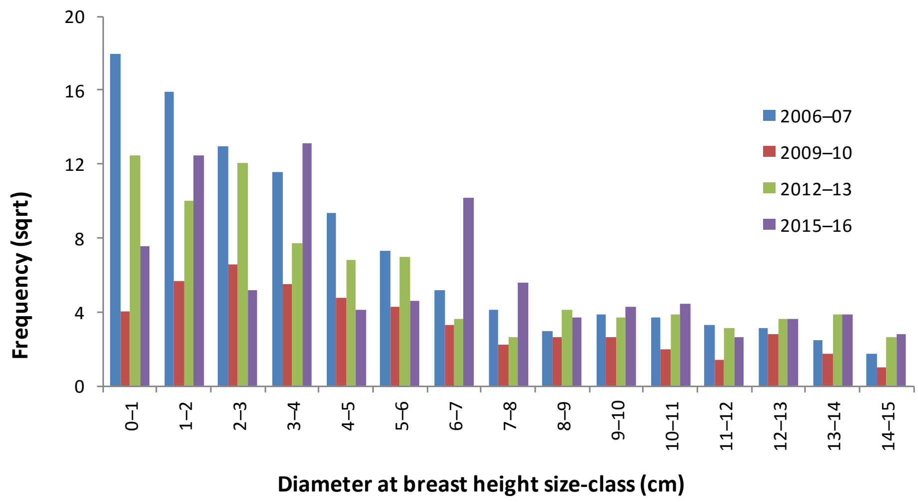


Figure 25. Size-class distribution of live riverine chenopod woodland trees (0–15 cm DBH) at Hattah Lakes; n(2006–07) = 1130, n(2009–10) = 210, n(2012–13) = 666, n(2015–16) = 681.

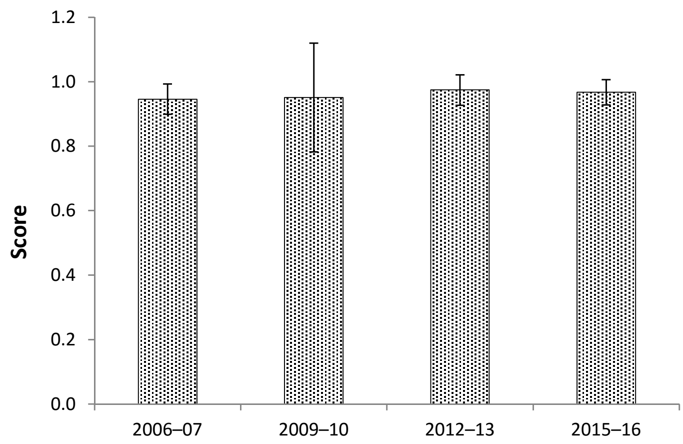


Figure 26. Population status index (± 95% CI) for riverine chenopod woodland calculated based on level of correlation with the reference 'inverse J-shaped' curve.

Population growth

Net population growth occurred in the riverine chenopod woodland WRC in 2015–16 (Figure 27). Since 2008–09 riverine chenopod woodland has recorded relatively low annual mortality (< 0.7%) and net population growth has been recorded annually since 2008–09. The high recruitment in 2008–09 for the riverine chenopod woodland is somewhat unusual. Almost 400 trees were assessed, and nearly all were juveniles, with only four adults and one 'recruit'. This is an artefact of spatial variability in the Black Box population.

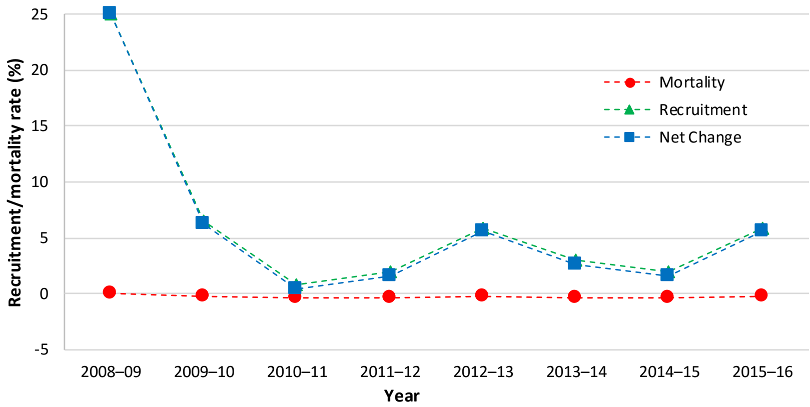


Figure 27. Net population growth for Black Box in riverine chenopod woodland, calculated as the difference between recruitment and mortality.

3.6 Discussion

At the end of the millennium drought (2010), Black Box condition at Hattah Lakes was generally poor. During 2010 and 2011 substantial improvement in tree condition occurred, which likely resulted from flooding and increased rainfall. Rainwater caused localised flooding and would have been readily used by the trees. Although no sites were directly influenced by floodwater at this time, the positive effects on groundwater level would have undoubtedly provided a longer-term benefit to trees, as Black Box are able to utilise groundwater when soil moisture is low (Jolly & Walker 1996).

Since 2011, below-average rainfall has persisted in the region; however the mean frequency of trees with a crown extent score ≥ 4 remained relatively steady. It may be that large-scale environmental watering in the icon site during 2013 and 2014 helped maintain tree condition. Although the watering did not directly inundate the majority of Black Box sites, it likely contributed to maintenance of a high groundwater level, such that it could be accessed and utilised by Black Box. While flooding (environmental or otherwise) may help maintain the condition of inundated Black Box and of trees adjacent to the floodwater, the condition of trees outside these regions is likely to decline if rainfall remains below average.

Since 2009–10, germination of Black Box has been relatively prolific in Black Box swampy woodland compared to riverine chenopod woodland. Location on the floodplain may account for this difference, with riverine chenopod woodland located higher on the floodplain, thus requiring larger floods than Black Box swampy woodland for inundation to occur. Black Box germination tends to be sporadic, with higher instances generally recorded following flooding or heavy rain (George et al. 2005). Natural flooding in 2010–11 and environmental watering in 2013 and 2014 resulted in inundation of some areas of Black Box swampy woodland but not riverine chenopod woodland.

The extent of inundation in 2014 was larger than in summer 2010–11. The increase in the number of newly germinated seedlings between 2012–13 and 2015–16 for Black Box swampy woodland may reflect the increase in the area of inundation between these floods.

The population of Black Box at Hattah Lakes is being maintained; in most years annual recruitment is either keeping pace with or exceeding mortality. Lack of representative Black Box swampy woodland sites for some years results in an incomplete data series, however the limited data that are available suggest that the population is being maintained. It is recommended that additional Black Box swampy woodland sites are added to the monitoring program so that this community is better represented across all future survey years.

As of 2015–16, it is deemed that the overall health of Black Box is being maintained above current targets, therefore the specific adopted objective *Sustainable populations of Black Box* is being met for Black Box communities at Hattah Lakes icon site.

4 Wetland vegetation communities

FIONA FREESTONE AND CHERIE CAMPBELL

4.1 Introduction

Wetlands of the Murray–Darling Basin can be permanent, temporary or ephemeral bodies of water of various depths. During overbank flooding, wetlands become connected to the river system and remain inundated as flood waters recede (Young 2001). During drawdown, wetlands provide a wet/dry ecotone that is high in species diversity compared with adjacent terrestrial and aquatic communities (Brock & Casanova 1997). Hydrology strongly influences the distribution and abundance of species found in the ecotone. River regulation has led to a reduction in the frequency, magnitude and duration of flooding events in the lower reaches of the Murray–Darling system (Rogers & Ralph 2011; Young 2001). If overbank floods continue to become less frequent and less variable, it is anticipated that the ecotone will shrink and wetland vegetation communities will be replaced by drought-tolerant species (Brock & Casanova 1997; Nicol & Weedon 2006; Roberts & Marston 2011; Rogers & Ralph 2011). The Living Murray program is a large-scale restoration project that attempts to ameliorate the negative effects of regulation on wetlands and the floodplain. Condition monitoring of wetland vegetation communities at the Hattah Lakes icon site has been undertaken since 2007–08. This chapter reports on the findings of these surveys over the last nine years.

4.2 Ecological objectives

The vision for the Hattah Lakes icon site is to:

> *Preserve and, where possible, enhance the biodiversity values of Hattah Lakes; and restore healthy examples of all original wetland and floodplain communities that represent the communities which would be expected under natural flow conditions (MDBA 2012c).*

The overarching ecological objective for vegetation at Hattah Lakes is:

> *Restore a mosaic of healthy wetland and floodplain communities to maintain the ecological character of the Ramsar site (MDBA 2012c).*

The Living Murray program is currently undergoing refinements that involve the development of operational objectives and the identification of suitable ecological indices that link back to the vision statement and ecological objective (Brown et al. 2015a; Robinson 2014a, b). At the time this report was compiled, operational objectives and indices for wetland vegetation were being developed and refined following a review of the condition monitoring program. As an interim measure for this monitoring period, this chapter reports on the established vision statement and overarching ecological objective by examining changes in wetland vegetation over time. This will be achieved by analysing species richness, the proportion of non-native species (i.e. weediness) and changes in community composition of wetland vegetation. This information may be used to assist in developing indices, or points of reference, for future condition monitoring reporting.

4.3 Methods

This section summarises the methods used to monitor and analyse species richness, proportion of non-native species and changes in wetland community composition over time. When condition monitoring began in 2007–08, wetland sites were established to represent the various sizes, shapes, commence-to-flow levels and vegetation communities that exist at Hattah Lakes. Sites were selected using satellite imagery, GIS models and layers such as the River Murray Floodplain Inundation Model (RiM-FIM) (Overton et al. 2006), and local knowledge. Nine wetlands have been surveyed annually since 2007–08. Additionally, a site at Lake Kramen was established and surveyed in May 2011, monitoring at Chalka Creek North began in 2011–12 and monitoring at Lake Bitterang began in 2012–13. All 12 wetland sites were surveyed during the 2012–13, 2013–14 and 2015–16 monitoring years (monitoring did not occur in 2014–15 due to changes in program funding).

Wetland vegetation survey procedures were based on those developed by Nicol and Weedon (2006). Vegetation was sampled at three or four permanently established transects per wetland. Along each transect, perpendicular to the transect line, quadrats (comprised of 15 x 1 m x 1 m cells) were surveyed at various elevations (depending on the depth of the wetland), from the base of the wetland up to the wetland edge (beyond the tree line). For the number of transects, quadrats and elevations at each individual wetland, refer to section 3.1 Part B of this report. Details of the survey methods used at Hattah Lakes can be found in section 6 of The Living Murray: Condition Monitoring Program design for Hattah Lakes (MDFRC 2011). It is possible that some plant species that occur at the Hattah Lakes icon site were not captured in the sampling method. Quadrat size and number (the use of 15 x 1 m x 1 m cells) was determined by Nicol and Weedon (2006) based on the results of species–area curves from the Chowilla icon site. Due to the likely similarity of plant communities between the Chowilla and Hattah Lakes icon sites, the same sampling intensity has been adopted for Hattah Lakes. However, given the size of the quadrats in comparison to the area surveyed, there may be some species with patchy distributions or low abundances not captured within the sampled quadrats. The seasonality of plant life cycles means that some species may not have been present at the time of the survey.

4.3.1 Wetland inundation state

The historic and current inundation state of each wetland provided context for data analysis. Over the last three years, environmental water has been delivered to Hattah Lakes via the pumping station at Chalka Creek. During surveys undertaken in 2015–16, 10 of the 12 wetlands were inundated (i.e. at least half of the quadrats at these wetlands were under water), Chalka Creek North was classified as intermittent-dry (i.e. all quadrats were dry, but the site had held water less than two years ago) and Lake Nip Nip was in the process of drawing down (i.e. less than half the quadrats there were under water) (Table 4.1). Prior to environmental watering in 2013, Lake Bitterang was last inundated by floodwater in ~1993–94. All other wetlands surveyed were inundated during the natural flood in 2010–11.

Table 4.1. Hydrological state of each wetland surveyed at Hattah Lakes in 2015–16. Key: inundated = at least half of the quadrats were inundated during the survey; drawdown = less than half the quadrats were inundated at the time of the survey; intermittent-dry = all quadrats dry, but wetland held water less than two years ago and may still display a vegetation response to inundation.

Wetland	Hydrological state 2015–16	Environmental water delivered
Lake Bitterang	Inundated	2013, 2014
Lake Boich	Inundated	2013, 2014
Lake Brockie	Inundated	2006, 2010, 2013, 2014, 2015
Lake Bulla	Inundated	2006, 2010, 2013, 2014, 2015
Chalka Creek	Inundated	2005, 2006, 2009, 2010, 2013, 2014, 2015
Chalka Creek North	Intermittent-dry	2013, 2014
Lake Hattah	Inundated	2006, 2009, 2010, 2013, 2014, 2015
Lake Kramen	Inundated	2011, 2014
Lake Little Hattah	Inundated	2005, 2006, 2009, 2010, 2013, 2014
Lake Mournpall	Inundated	2006, 2009, 2010, 2013, 2014, 2015
Lake Nip Nip	Drawdown	2013, 2014
Lake Yerang	Inundated	2006, 2009, 2010, 2013, 2014, 2015

4.3.2 Plant species classification

Plant species identification

Plants were identified using the Flora of Victoria Volumes 2 and 3 (Walsh & Entwisle 1994, 1996) and the online version (http://data.rbg.vic.gov.au/vicflora/), Flora of New South Wales Volumes 1–4 (Harden 1992, 1993, 2000, 2002) and the online version (http://plantnet.rbgsyd.nsw.gov.au) and information from field guides (Cunningham et al. 1992; Sainty & Jacobs 1981). As the study area is in Victoria, scientific and common names follow those used in the Flora of Victoria (online). Where species are not recognised for Victoria, scientific and common names follow the Flora of New South Wales (published and online).

The conservation significance of plant species was determined using listings in the Flora of Victoria (online version). Non-native species are identified with an asterisk (*) throughout this report.

Some plant species samples could only be identified to genus or family level, or were unidentifiable due to insufficient plant material. It was not possible to determine if these particular species were the same as those recorded in previous years, which can affect between-year comparisons at the species level. Using plant functional groups ameliorates this to a large extent.

Functional groups

The plant functional group (FG) approach has been widely used to assist in interpreting and predicting change in plant community function and dynamics in relation to a disturbance (Brock & Casanova 1997). Minor changes in species composition or inconsistencies in taxonomic resolution may affect between-year comparisons and the ability to detect ecologically significant changes in community structure. The use of functional groups helps to minimise these inconsistencies by detecting changes in community structure based on plant responses to water regimes.

Plant species recorded in surveys at Hattah Lakes were classified into functional groups (Table 4.2). Functional group classification for each species is provided in section 3 in Part B of this report. The classification of plant species into these groups is based largely on Brock and Casanova (1997) and Reid and Quinn (2004). Species that were not classified in either of these studies were assigned to functional groups based on field observations and information in the Flora of Victoria (online version) and Cunningham *et al.* (1992). An additional floating (F) functional group was added to identify species not attached to the substrate. Functional group T (instead of Tdr or Tda) and A (instead of Ate, Atl, Arf or Arp) were assigned where species were identified to genus or family level only.

Table 4.2. Functional groups used to classify species recorded in wetlands at the Hattah Lakes.

FG	Description
S	Aquatic submerged species (established plants do not tolerate drying).
F	Aquatic floating, unattached species (established plants do not tolerate drying).
A	Amphibious species (plants that tolerate both flooding and drying).
Ate	Amphibious, fluctuation-tolerant, emergent species which are mostly monocotyledons (emergent plants that tolerate wetting and drying).
Atl	Amphibious, fluctuation-tolerant, emergent species which are dicotyledons and require damp conditions (low-growing plants that tolerate wetting and drying).
Atw	Amphibious, fluctuation-tolerant, emergent plants which are woody (trees and shrubs that tolerate wetting and drying).
Arf	Amphibious, fluctuation-responder, floating species which have floating leaves in their aquatic phases and also grow stranded on damp ground.
Arp	Amphibious, fluctuation-responder, floating species, with various growth characteristics, that feature morphological plasticity in response to water level fluctuations.
T	Terrestrial species (plants that do not tolerate flooding).
Tdr	Terrestrial species that typically occur in dry habitats.
Tda	Terrestrial species that typically occur in damp habitats.

Data analysis

Analyses of changes in wetland vegetation community composition over time were carried out in PRIMER (Version 6), with the PERMANOVA+ add-in (Anderson et al. 2008). Water regime classes were assigned to each wetland for analysis (Table 4.3).

Table 4.3. Water regime class for each wetland at Hattah Lakes (Ecological Associates 2007).

Water regime class	Wetlands
Semi-permanent wetlands	Lakes Hattah, Bulla, Mournpall, Brockie
Persistent temporary wetlands	Lakes Little Hattah, Boich, Nip Nip, Yerang, Bitterang
Episodic wetlands	Lake Kramen
Anabranches	Chalka Creek, Chalka Creek North

Functional group abundance data (e.g. a score from between 0 to 15, based on the number of cells surveyed per quadrat) was averaged for each transect, in each wetland, for each year. The Bray-Curtis resemblance measure was then applied to generate a resemblance matrix, with a dummy value of one added to account for quadrats that had no species (i.e. either inundated or bare ground).

Permutational multivariate analysis of variance (PERMANOVA) was used to analyse functional-group abundance data (averaged for each transect in each wetland, per year), with 'monitoring year' and 'WRC' as fixed factors. Where significant interactions were detected between factors, the main-effects test was followed by pairwise comparisons. The PERMANOVA analyses were performed using 9999 permutations under a reduced model for two-way PERMANOVAs, as recommended by Anderson *et al.* (2008). Significance was reported using Monte Carlo *P*-values as, for some pairwise comparisons, there were insufficient units to enable calculation of a rigorous test statistic using permutation (Anderson 2005). A multi-dimensional scaling (MDS) ordination plot was used to display overall differences in functional group abundance and composition between both WRCs and monitoring years. For display purposes, the MDS ordination is based on mean functional-group composition and abundance data per WRC for each year.

4.4 Results

4.4.1 Species richness

A total of 74 plant species were recorded in 2015–16 across all wetlands at Hattah Lakes (Table 4.4). Across the eight survey years, the greatest species richness was recorded in 2012–13 as water receded following natural flooding in 2010–11. At the original nine sites, species richness was greatest in 2008–09 and 2009–10 possibly as a result of inundation at some of these sites during survey between 2010 and 2012. It is anticipated that species richness at these sites will increase in future surveys, as wetlands continue to drawdown. The lowest number of species was recorded in 2013–14 in association with environmental flows, when species from the floating functional group dominated the plant community. The total number of native and non-native species recorded in each year is displayed in Figure 28. For a comprehensive list of all plant species recorded at each site, refer to section 3 in Part B of this report.

Table 4.4. Number of plant species recorded at all wetlands at Hattah Lakes in all monitoring years.

Monitoring year	2007–08	2008–09	2009–10	2010–11	2011–12	2012–13	2013–14	2015–16
Number of sites surveyed	9	9	9	10	11	12	12	12
Total species recorded	77	86	86	71	107	118	52	74
Total species recorded at original sites*	77	86	86	37	68	80	15	50

*Includes only the nine sites that were originally surveyed in 2007–08.

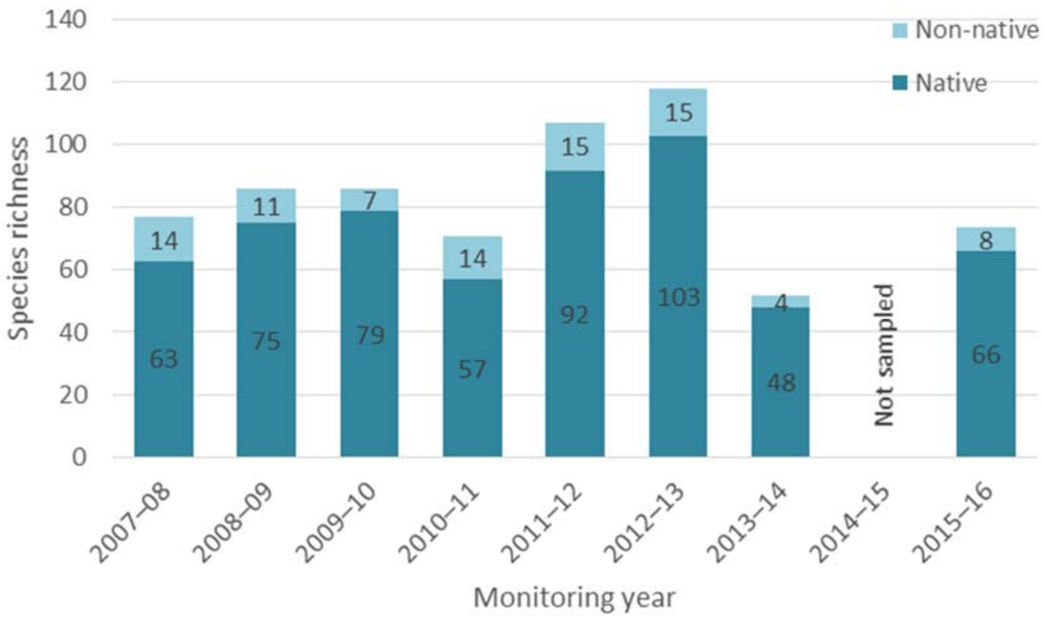


Figure 28. Number of native and non-native plant species recorded across all wetlands at Hattah Lakes in all monitoring years (*n* = 9 in 2007–08, 2008–09, 2009–10; *n* = 10 in 2010–11; *n* = 11 in 2011–12; *n* = 12 in 2012–13, 2013–14, 2015–16).

The most common wetland species recorded at Hattah Lakes in 2015–16 are listed in Table 4.5 Common species are defined as those where total abundance (number of survey quadrats in which the species occurred) was greater than 100 (out of a possible 3195, being the number of 1 m x 1 m cells surveyed across all wetlands).

Table 4.5. The most common plant species recorded in wetlands at the Hattah Lakes icon site in 2015–16. Key: Atl = amphibious monocotyledons, Arp = amphibious floating plants, Tda = terrestrial plants preferring damp habitats.

Scientific name	**Common name**	**Functional group**	**Abundance (out of 3195)**
Myriophyllum verrucosum	Red Water-milfoil	Arp	682
Stemodia florulenta	Blue Rod	Tda	246
Glycyrrhiza acanthocarpa	Southern Liquorice	Tda	219
Elatine gratioloides	Waterwort	Arp	156
Sphaeromorphaea littoralis	Spreading Nut-heads	Tda	150
Centipeda cunninghamii	Common Sneezeweed	Atl	142
Helichrysum luteoalbum	Jersey Cudweed	Tda	108

Three species recorded in 2015–16 had not been previously recorded at the Hattah Lakes wetlands during TLM condition monitoring: *Gratiola pubescens* (FG = Tda); *Rumex crystallinus* (Glistening Dock, FG = Tda), which is listed as vulnerable in Victoria (see section 4.4.3 for more details about rare and threatened species); and *Juncus subsecundus* (Finger Rush; FG = Ate).

4.4.2 Non-native species

Because different numbers of wetlands were surveyed in different years, non-native species were assessed as a proportion of species abundance (Figure 29). The proportion of non-native species was highest in 2010–11. Non-native species were less than 15% of the total abundance in each monitoring year.

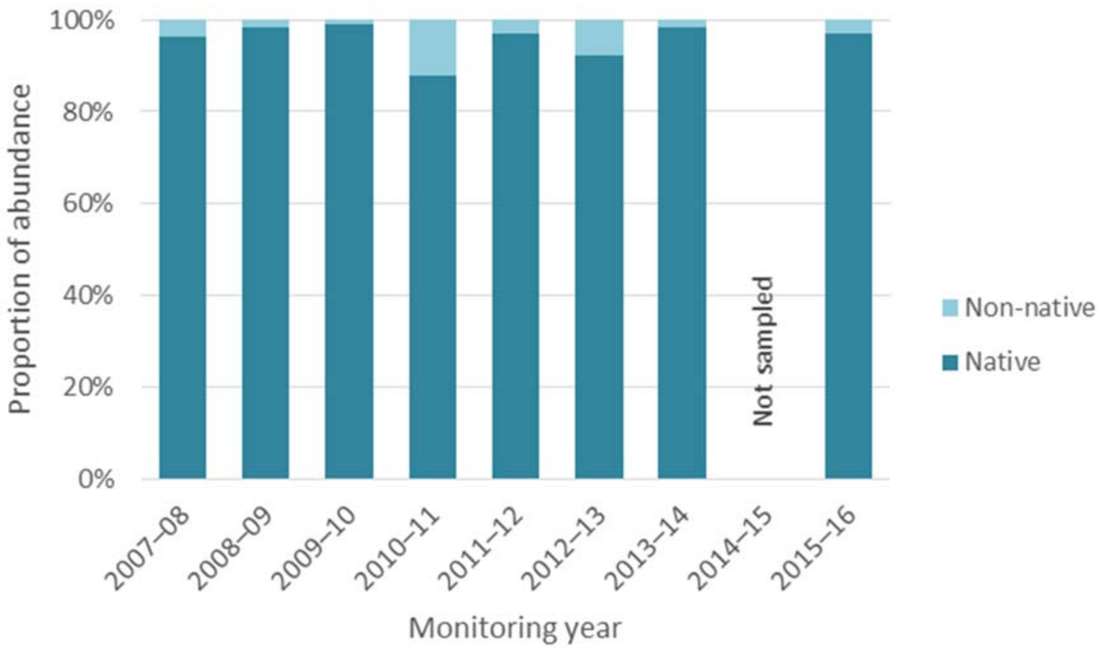


Figure 29. Proportion of non-native species abundance in wetlands at Hattah Lakes in all monitoring years (*n* = 9 in 2007–08, 2008–09, 2009–10; *n* = 10 in 2010–11; *n* = 11 in 2011–12; *n* = 12 in 2012–13, 2013–14, 2015–16).

4.4.3 Rare or threatened species

Of the 74 species recorded in 2015–16, four were listed as having conservation significance in Victoria (Table 4.6). Three of these species were recorded at Lake Bitterang, a long-dry wetland which was drawing down following the delivery of environmental water in 2013 and 2014. Glistening Dock, which was recorded at Lakes Bitterang and Nip Nip, had not previously been recorded at the Hattah Lakes wetlands during TLM condition monitoring.

Surveys undertaken for TLM do not specifically target species with conservation significance. The abundance count for each species (out of a possible 3195, being the number of 1 m x 1 m cells surveyed at the Hattah Lakes wetlands) is included in Table 4.6. For comparative purposes, the most abundant species recorded at the Hattah Lakes wetlands in 2015–16 was Red Water-milfoil (*Myriophyllum verrucosum*), which had an abundance count of 682. More information on rare or threatened species recorded at the Hattah Lakes wetlands in previous monitoring years can be found in section 3 in Part B of this report.

Table 4.6. Rare or threatened plant species recorded at wetland sites at Hattah Lakes in 2015–16. Key: FG = functional group: Ate = amphibious monocotyledons, Tda = terrestrial species that typically occur in damp habitats, Tdr = drought-tolerant species; CS = conservation status in Victoria: k = poorly known, r = rare, v = vulnerable. ^Habitat preference is from Cunningham et al. (1992) and Harden (1992, 1993, 2000, 2002) and notes from previous TLM records.

FG	Scientific name	Common name	Family	CS	Recorded at	Habitat preference^	Abundance (out of 3195)
Tdr	*Calotis cuneifolia*	Blue Burr-daisy	Asteraceae	r	Lake Bitterang	Grows mostly in sandy and red clay loam soils in a wide range of plant communities. Recorded in previous monitoring years at Lakes Bitterang, Kramen and Yerang.	12
Ate	*Isolepis australiensis*	Inland Club-sedge	Cyperaceae	k	Lake Boich	Grows in seasonally wet situations. Recorded previously at Lake Mournpall in 2012–13.	1
Tda	*Rumex crystallinus*	Glistening Dock	Polygonaceae	v	Lake Bitterang Lake Nip Nip	Generally grows in damp or low-lying areas, often around the perimeter of ephemeral lakes, on soil from which water has receded. Recorded for the first time at TLM Hattah wetland sites during 2015–16 surveys.	18
Tdr	*Swainsona microphylla*	Small-leaf Swainson-pea	Fabaceae	r	Lake Bitterang	Grows mostly in light soils on sand hills.	1

4.4.4 Functional groups

Functional group abundance data are displayed for wetlands surveyed over four monitoring years (Figure 30). Monitoring years were chosen to display changes in community composition according to inundation events.

The first year of monitoring occurred in 2007–08, during the millennium drought (Figure 30 (a)). Species from amphibious functional groups (e.g. A, Arp, Ate and Atw) were recorded as a result of environmental water having been delivered to the lower lakes in 2005 and 2006 (e.g. Lakes Brockie, Bulla, Hattah, Little Hattah, Mournpall, Yerang and Chalka Creek).

The natural flood in 2010–11 inundated nine wetlands and Lake Kramen was inundated by environmental water just prior to the survey (Figure 30 (b)). Community composition in 2010–11 was dominated by species from the floating (F) functional group.

In 2013–14, 11 wetlands were inundated following delivery of environmental water in 2013 (Figure 30 (c)). Due to the extent of inundation, the overall community composition was similar to that recorded during the natural flood (2010–11).

The majority of lakes surveyed remained inundated in 2015–16, following consecutive years of environmental watering (Figure 30 (d)). In this monitoring year, species were recorded from an array of functional groups, with the highest abundance recorded in Arp (amphibious, floating species, e.g. Red Water-milfoil) and Tda (terrestrial species occurring in damp habitats).

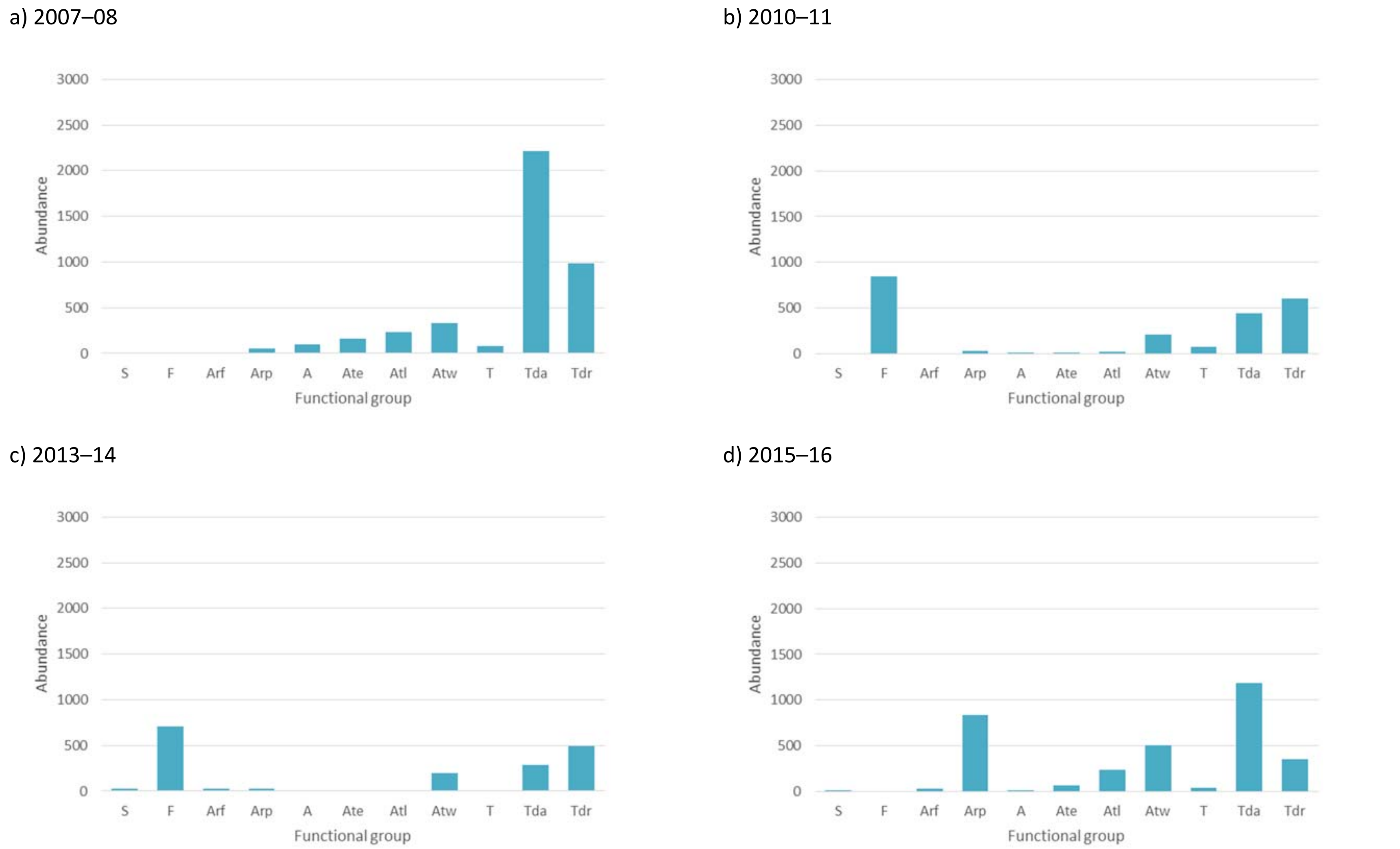


Figure 30. Functional group abundance data for all wetlands over four monitoring years at Hattah Lakes (n = 9 in 2007–08; n = 10 in 2010–11; n = 12 in 2013–14 and 2015–16).

4.4.5 Community composition

Community composition differs between WRCs and monitoring years. As there was a significant interaction between year and WRC (*P* = 0.0001), PERMANOVA pairwise tests were undertaken, based on mean composition and abundance data per transect, per site, per year, for each WRC (Table 4.7). In semi-permanent, persistent temporary and episodic wetlands, a statistically significant difference was found between 2015–16 and every other monitoring year. In the anabranch WRC, there was no statistically significant difference between the 2015–16 and 2007–08 monitoring years, nor between 2015–16 and 2008–09.

Table 4.7. PERMANOVA (pairwise) test results showing differences in functional group composition and/or abundance between years for each WRC. Significance has been reported using Monte Carlo *P*-values and statistically significant effects are indicated in bold (α ≤ 0.05).

WRC	2015–16 vs 2007–08	2015–16 vs 2008–09	2015–16 vs 2009–10	2015–16 vs 2010–11	2015–16 vs 2011–12	2015–16 vs 2012–13	2015–16 vs 2013–14
Anabranches	0.3812	0.3704	**0.0004**	**0.0040**	**0.0245**	**0.0187**	**0.0047**
Semi-permanent wetlands	**0.0001**	**0.0001**	**0.0001**	**0.0003**	**0.0041**	**0.0001**	**0.0002**
Persistent temporary wetlands	**0.0118**	**0.0064**	**0.0044**	**0.0001**	**0.0053**	**0.0086**	**0.0001**
Episodic wetlands	NA	NA	NA	**0.0012**	**0.0005**	**0.0003**	**0.0003**

The MDS ordination (Figure 31), displays changes in functional group composition and abundance over time. Mean composition and abundance data per WRC for each year were used for display purposes. Data from the anabranch WRC in 2009–10 differs from data in all other WRCs in all years (Figure 31 (a)). This is likely due to the high abundance of species from the submerged (S) and amphibious floating (Arf) functional groups recorded at Chalka Creek in 2009–10, as indicated by the vector.

The inset in Figure 31 (b) is a close-up of the relationship between all monitoring years in each of the WRCs. The absence of strong trends in the data is likely a result of wetlands being inundated by environmental water and/or flooding, or vegetation responses to the recession of flows, during most monitoring years. That is, in each WRC in each monitoring year, species from an array of amphibious and terrestrial functional groups were recorded, supported by the environmental watering and flooding that has occurred at Hattah Lakes over the last decade.

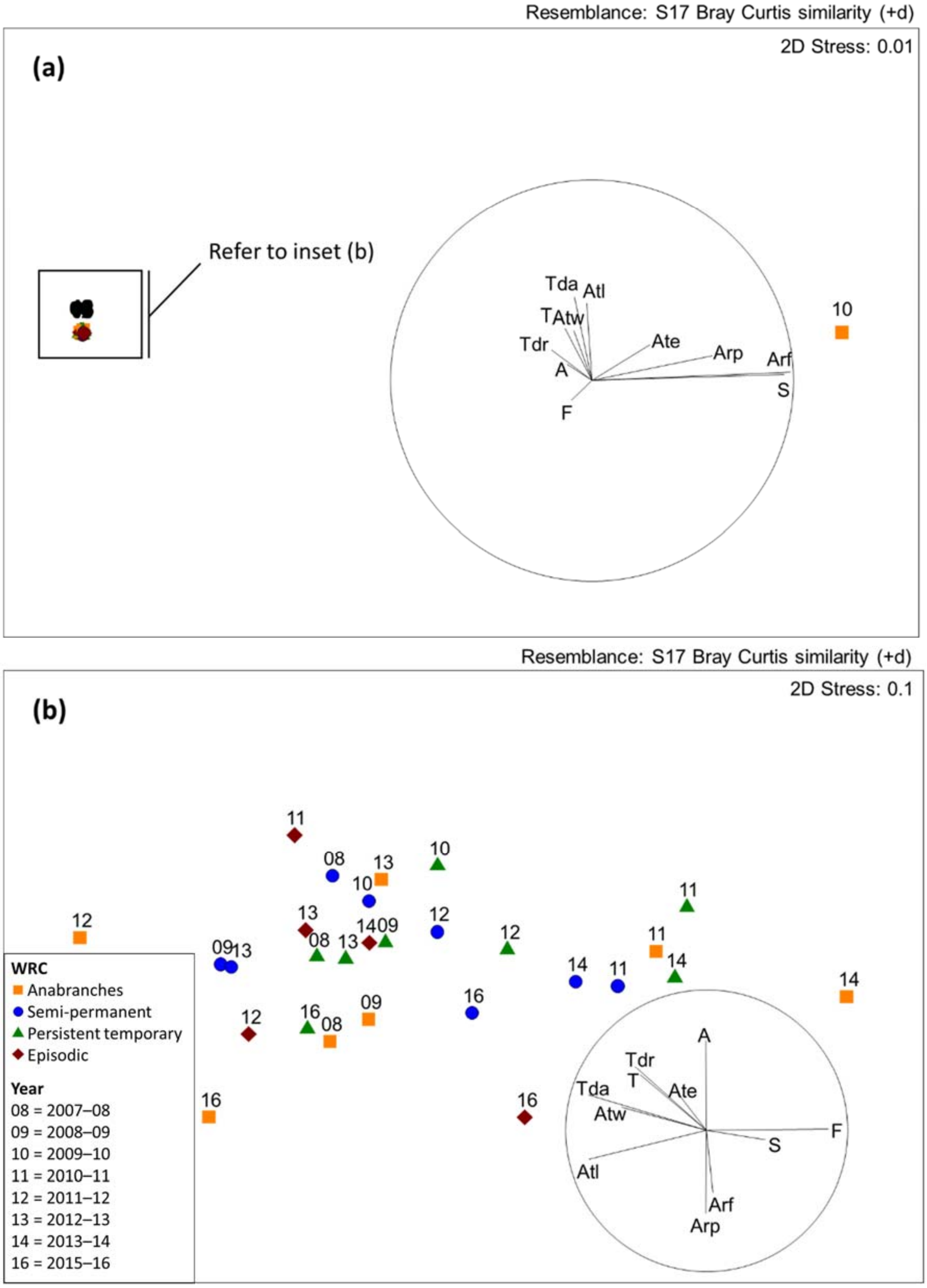


Figure 31. Differences in vegetation functional group composition at Hattah Lakes between WRCs and monitoring years (a). Inset of (a) displayed in (b).

4.5 Discussion

4.5.1 Species richness and abundance

Favourable environmental conditions have benefited the wetland water-responsive plant community at Hattah Lakes. The majority of wetlands surveyed were inundated in 2015–16 through delivery of environmental water over three consecutive years (2013, 2014 and 2015). A total of 74 plant species were recorded with the most abundant being Red Water-milfoil, an amphibious floating species (functional group Arp). The seven most common plants recorded are considered water-responsive (e.g. species from functional groups Atl, Arp and Tda). Repeated inundation is likely to facilitate vegetative growth (e.g. lateral growth from underground root systems) or completion of plant life cycles (e.g. the replenishment of seedbanks), enabling species to respond with greater abundance following subsequent flows. It is likely that these wetland vegetation communities at Hattah Lakes have benefited from repeated inundation since flooding in 2010–11.

Of the 74 species recorded in 2015–16, four were listed as having conservation significance in Victoria. Three of these species were recorded at Lake Bitterang, a wetland that was long-dry (approximately 20 years) prior to the delivery of environmental water in 2013. While targeted rare plant surveys are not undertaken as part of TLM condition monitoring, data suggests that delivery of environmental water on multiple occasions has improved habitat for flow-dependent rare species at Hattah Lakes. Targeted surveys for flow-dependent rare plant species are recommended.

In 2015–16, three species were recorded for the first time in nine years of TLM condition monitoring at the Hattah Lakes wetlands. Though new to this dataset, two of these species had been previously recorded at Hattah: Finger Rush, previously recorded during floodplain condition monitoring (Henderson et al. 2012), and Glistening Dock (EPA & MDFRC 2008). After almost a decade of condition monitoring, it is interesting that new species are still being added to the TLM wetland dataset. It is likely that this is a reflection of the variety of habitats surveyed each year (e.g. different wet/dry status at each wetland), the timing of surveys in relation to inundation (e.g. recently filled, length of time filled) and the transient nature of wetland plant communities (e.g. water-responsive species generally only live for about one year).

The greatest number of species and abundance overall was recorded in 2012–13, largely due to the drawdown of wetlands following the natural flood in 2010–11. In the most recent survey (2015–16), 11 of the 12 monitored wetlands still held water (10 inundated, 1 in drawdown). It is anticipated that the composition and abundance of species in amphibious and terrestrial damp functional groups will increase in future surveys as wetlands continue to draw down.

In 2015–16, non-native species were less than 10% of the proportional species abundance. The proportion was less than 15% in all monitoring years. The greatest proportional abundance of non-native species was recorded during the flood year (14%), which is attributed to above-average rainfall prior to the survey (2010–11) and lower overall records of vegetation abundance due to the extent of inundation. Though the majority of wetlands were inundated when surveyed in 2010–11, Lake Nip Nip had only just commenced filling with natural floodwater and environmental water that was delivered to Lake Kramen had not reached its highest bank. The majority of non-native species were recorded in the exposed elevations that were surveyed at these two wetlands. For wetland ecosystems in Victoria, the average relative non-native species cover is 15–20% and relative non-native species richness is ~35% (Catford & Jansson 2014). In all years, the relative proportion of non-native species abundance and the relative non-native species richness recorded at Hattah was below the average for Victoria.

4.5.2 Community composition based on functional groups

The use of functional groups is a widely accepted method of interpreting changes in plant communities in relation to disturbance, while minimising the effects of changes in species composition or inconsistencies in taxonomic classification (Brock & Casanova 1997; Campbell et al. 2014). Functional groups demonstrate the impact of inundation on community composition. Based on functional group data, community composition in the anabranch WRC in the 2009–10 monitoring year was dramatically different to every other WRC in every other monitoring year. This is a result of the unusually high abundance of species from the S (submerged) and Arp and Arf (amphibious floating) functional groups recorded at Chalka Creek in 2009–10 following environmental watering (e.g. S – Eel Grass *Vallisneria australis;* Arp – Waterwort *Elatine gratioloides*, Small Mud-mat *Glossostigma elatinoides* and Red Water-milfoil; Arf – Swamp Lily *Ottelia ovalifolia* subsp*. ovalifolia* and Furrowed Pondweed *Potamogeton sulcatus*).

In all other monitoring years, community composition at Hattah Lakes was made up of species from an array of amphibious and terrestrial functional groups in all WRCs. Although there were statistically significant differences between all WRCs in most monitoring years, there were no strong trends correlated with drought or flood years. This is likely a result of some wetlands being inundated by environmental water during drought years (2007–08 to 2009–10 surveys), a response to floodwater recession (2011–12 and 2012–13 surveys) and environmental water delivered over the last three years (2013, 2014 and 2015). Variable wetting and drying regimes result in changes in plant community composition over time and across the landscape (Capon 2005; James et al. 2007; Warwick & Brock 2003). An increase in the diversity of wetland habitats is associated with an increase in species richness and diversity (Stein et al. 2014; Thoms et al. 2006). In each monitoring year, wetlands at Hattah Lakes were in various states of inundation (e.g. inundated, drawdown, long-dry), providing a diverse mosaic of habitats and supporting a variety of vegetation communities. Therefore, the overarching ecological objective to restore a mosaic of healthy wetland communities at Hattah Lakes is being addressed.

4.5.3 Recommendations

Consecutive environmental flows appear to be supporting flow-dependent rare plants. There is limited information about these species, largely because of their ephemeral nature. Threatened species recovery plans and action plans are designed to determine how many populations of these species exist and map where those populations are located. This is problematic for flow-dependent rare plants that are short-lived (e.g. often less than 12 months) and emerge only following an episodic event (e.g. after inundation). These species will likely benefit from environmental watering. Targeted surveys (timed to coincide with the drawdown of wetlands following the delivery of environmental water) are recommended and would provide valuable information for the management of flow-dependent rare plants.

Three species of conservation significance in Victoria were recorded at Lake Bitterang in 2015–16. Lake Bitterang was dry for approximately 20 years prior to environmental water being delivered in 2013 and 2014. Further monitoring over the next couple of years as the lake continues to draw down will provide valuable information as to the continued vegetation community response to water over time.

It is anticipated that the composition and abundance of species in amphibious and terrestrial damp functional groups will increase in future surveys as wetlands draw down. Comparisons of ongoing condition monitoring data (as wetlands draw down) to vegetation responses following natural floods (e.g. 2011–12 and 2012–13/2015–16) will be useful in guiding the delivery of future environmental flows.

4.5.4 Summary

Key points from condition monitoring of wetland vegetation at Hattah Lakes in 2015–16 are:

- Water-responsive plant species at Hattah Lakes in 2015–16 have benefited from favourable environmental conditions. The lakes were inundated by floodwater in 2010–11 and environmental water has been delivered over three consecutive years since then (2013–2015).
- Red Water-milfoil was the most abundant species recorded at the Hattah Lakes wetlands in 2015–16 (Figure 32 and Figure 33).
- The seven most abundant plant species recorded in 2015–16 were water-responsive.
- A diversity of habitat types have been supported within the Hattah Lakes wetlands in 2015–16 (e.g. inundation to different depths, recent drawdown and damp mud, and flow recession in the last year or two). This habitat mosaic supports species diversity and abundance.

Figure 32. A carpet of Red Water-milfoil as Lake Boich draws down (F Freestone, December 2015).

Figure 33. Abundant submerged and emergent Red Water-milfoil at Lake Little Hattah (F Freestone, December 2015).

5 Floodplain vegetation communities

FIONA FREESTONE AND CHERIE CAMPBELL

5.1 Introduction

Floodplains have been defined as areas of relatively flat land that are inundated when adjacent rivers overflow their banks during a flood (Young 2001). In arid landscapes, floodplains provide critical aquatic and riparian habitat for flora and fauna that are both reliant on and tolerant of flooding (Rogers & Ralph 2011). The distribution and abundance of floodplain vegetation is strongly influenced by hydrology and many species have adapted to depend on flooding (Brock & Casanova 1997). River regulation has led to a reduction in the frequency, magnitude and duration of flooding in the lower reaches of the Murray–Darling River system (Rogers & Ralph 2011; Young 2001). If floods continue to become less frequent and less variable, it is anticipated that these floodplain vegetation communities will be replaced by drought-tolerant species in the long term (Nicol & Weedon 2006; Roberts & Marston 2011; Rogers & Ralph 2011; Young 2001). The Living Murray program is a large-scale restoration project that attempts to ameliorate the negative effects of regulation on wetlands and the floodplain. Condition monitoring of floodplain vegetation communities at the Hattah Lakes icon site has been undertaken since 2007–08. This chapter reports on the findings of these surveys over the last nine years.

5.2 Ecological objectives

The ecological objectives used for floodplain vegetation communities are consistent with those used for wetland vegetation communities described in section 4.2.

5.3 Methods

This section summarises the methods used to monitor and analyse species richness, proportion of non-native species and changes in floodplain community composition over time. Annual condition monitoring was carried out at 17 sites at Hattah Lakes since 2007–08 (MDFRC 2011) (excluding 2014–15, when no data was collected due to changes in funding). These sites were established to represent the various vegetation communities and watering regimes that exist on the floodplain at Hattah Lakes. Water regime classes were developed to assist in classifying the floodplain into areas with common ecological characteristics and watering requirements (Ecological Associates 2007). Water regime classes include vegetation communities represented by species listed in specific EVCs. The EVCs, WRCs and vegetation communities were assigned to each floodplain site based on Geographic Information System (GIS) layers and on-ground site assessments (

Table 5.1). Irregularities may exist between digitised and ground-truthed boundaries in some instances. While these classifications are useful on a broad scale, it is important to remember that specific floodplain monitoring sites may not fit neatly into these classifications. Discrepancies were addressed on a case-by-case basis.

Each year, surveys were conducted between December and March, following the methods described in The Living Murray: Condition Monitoring Program design for Hattah Lakes (MDFRC 2011). Possible limitations associated with the use of these methods are described in section 4.3.

Table 5.1. The WRCs related to floodplain vegetation communities at Hattah Lakes (Ecological Associates 2007; Scholz et al. 2007a). Key: RRG = River Red Gum, BB = Black Box.

Sites	EVC	WRC	Vegetation community	Characteristics of classes
H2A H2B H3A H3B	EVC 106: Grassy riverine forest	Red Gum forest	RRG	Found only in areas subject to the most frequent flooding regimes. This WRC is (historically) subject to inundation in nearly all years and is characterised by a closed canopy of tall River Red Gum.
H5A H6A	EVC 813: Intermittent swampy woodland	Fringing Red Gum woodland	RRG	Occurs mainly in floodplain areas immediately surrounding wetlands and along water courses that are (historically) inundated by peaks in river flow during most years.
H1A H1B H4A H4B H5B	EVC 295: Riverine grassy woodland	Red Gum with flood-tolerant understorey	RRG	Represents the driest habitat for River Red Gum communities, in which Black Box may also be present. Historically, floods in this WRC are intermittent and brief. The understorey is dominated by terrestrial species that may suffer during flood, but which are favoured by the longer dry phase of this WRC.
H1C H2C H3C H5C H6B H6C	EVC 103: Riverine chenopod woodland EVC 823: Lignum swampy woodland	Black Box woodland	BB	Occurs in the least-frequently inundated areas of the floodplain. This WRC is dominated by Black Box with a diverse shrubby understorey of chenopods or Lignum.

5.3.1 *Floodplain inundation state*

The historic and current inundation state of each floodplain site provided context for data analysis. Since flooding in summer 2010–11, environmental water has been delivered to the lakes in spring 2013, 2014 and 2015, inundating floodplains within close proximity to the lakes (Table 5.2). Two sites were also influenced by environmental water pumped to the lakes between 2005 and 2010. In 2015–16, one floodplain site was inundated during monitoring (i.e. at least half of the four quadrats per site were inundated) and five sites were intermittent-dry (i.e. dry during this survey, but held water less than two years ago and may still show some vegetation response to inundation). The remaining 12 sites were considered long-dry as they had been dry for at least two monitoring years. Four of the six sites within Black Box communities have now not been inundated for approximately 20+ years.

Table 5.2. Vegetation community and hydrological state of each floodplain site in 2015–16. Key: RRG = River Red Gum, BB = Black Box.

Vegetation community	Site	Hydrological state 2015–16	Last inundated
BB	H1C	Long-dry	~1993–94
BB	H2C	Long-dry	~1993–94
BB	H3C	Long-dry	~1993–94
BB	H5C	Intermittent-dry (environmental water)	2014*
BB	H6B	Intermittent-dry (environmental water)	2014
BB	H6C	Long-dry	~1993–94
RRG	H1A	Long-dry	2010–11
RRG	H1B	Long-dry	2010–11
RRG	H2A	Long-dry	2010–11
RRG	H2B	Long-dry	2010–11
RRG	H3A	Long-dry	2010–11
RRG	H3B	Long-dry	2010–11
RRG	H4A	Intermittent-dry (environmental water)	2014
RRG	H4B	Long-dry	2010–11
RRG	H5A	Inundated (environmental water)	2015
RRG	H5B	Intermittent-dry (environmental water)	2014
RRG	H6A	Intermittent-dry (environmental water)	2015
NA	H4C^	Long-dry	NA

*Partially inundated (three out of four transects were inundated in 2013–14, one exposed).
^Located on a sand hill in EVC 824 (Woorinen Mallee) with no associated WRC.

Site H4C is located on a sand hill in EVC 824 (Woorinen Mallee) with no associated WRC. This site was surveyed in this monitoring year (species lists can be found in section 4 in Part B of this report), however it is not anticipated or desirable that sand hills be inundated through environmental watering. As such the data from this site was omitted from analysis in this report and it is recommended that surveying of this site be discontinued in future monitoring years.

5.3.2 Plant species classification

The methods used to identify plant species and the use of plant functional groups were as described in section 4.3.2.

5.3.3 Data analysis

Analyses of change in floodplain vegetation community composition over time were carried out in PRIMER (Version 6), with the PERMANOVA+ add-in (Anderson et al. 2008). The Bray-Curtis resemblance measure was applied to functional-group abundance data to generate a resemblance matrix, with a dummy value of one added to account for quadrats that had no species (i.e. either inundated or bare ground).

To analyse functional group abundance data, PERMANOVA was used, with 'monitoring year' and 'vegetation community' as fixed factors. Where significant interactions were detected between factors, the main-effect test was followed by pairwise comparisons. The PERMANOVA analyses were performed using 9999 permutations under a reduced model for two-way PERMANOVAs, as recommended by Anderson et al. (2008). For some pairwise comparisons, there were insufficient units to enable calculation of a rigorous test statistic using permutation. In these cases, significance was reported using Monte Carlo *P*-values (Anderson 2005). An MDS ordination plot was used to display overall differences in functional group abundance and composition between both vegetation community and monitoring years. For display purposes the MDS ordination is based on functional-group abundance data averaged for each year within each vegetation community (i.e. River Red Gum = 11 understorey sites and Black Box = 6 understorey sites).

5.4 Results

5.4.1 Species richness

A total of 113 plant species were recorded in 2015–16 across all floodplain sites at Hattah Lakes (Table 5.3). This is the second highest record of species richness for the nine monitoring years. The highest species richness was recorded in 2011–12 in response to natural flooding, which occurred in the summer of 2010–11. For a comprehensive list of all plant species recorded, refer to section 4 in Part B of this report.

Table 5.3. Number of sites surveyed and plant species recorded at floodplain understorey vegetation sites at Hattah Lakes in all monitoring years.

Monitoring year	**2007–08**	**2008–09**	**2009–10**	**2010–11**	**2011–12**	**2012–13**	**2013–14**	**2015–16**
Number of sites surveyed	17	17	17	14	17	17	17	17
Total species recorded	60	61	103	79	131	102	100	113

The most common floodplain species recorded at Hattah Lakes in 2015–16 are listed in Table 5.4. Common species are defined as those that had an abundance count greater than 100 (out of a possible 1020, being the number of 1 m x 1 m cells surveyed across the floodplain).

Table 5.4. The most common plant species recorded on the Hattah Lakes floodplain in 2015–16. Key: Atl = amphibious herbs, Tda = terrestrial species preferring damp habitats, Tdr = drought-tolerant species.

Scientific name	**Common name**	**Functional group**	**Abundance (out of 1020)**
Enchylaena tomentosa var. *tomentosa*	Ruby Saltbush	Tdr	194
Alternanthera denticulata	Lesser Joyweed	Tda	182
Centipeda cunninghamii	Common Sneezeweed	Atl	114
Euphorbia dallachyana	Flat Spurge	Tdr	102

In 2015–16, ten species were recorded that had not previously been recorded at Hattah Lakes floodplain sites in the last nine years of TLM condition monitoring (Table 5.5).

Of these newly recorded species, Austral Mudwort (*Limosella australis*), Red Water-milfoil and Eel Grass are water-responsive species recorded at floodplain sites that were either inundated or very recently drawn down following environmental flows delivered in 2015. Red Water-milfoil was also the most abundant species recorded at Hattah wetlands in 2015–16 (see section 4.4.1). Jerry-jerry (*Ammannia multiflora*) (Figure 34) and Glistening Dock are both listed as vulnerable in Victoria.

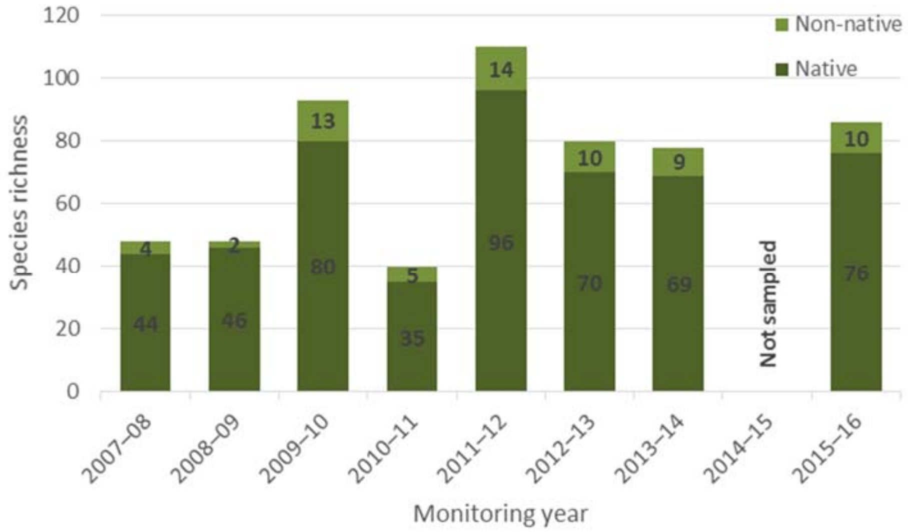
Table 5.5. Species recorded at Hattah Lakes in 2015–16 that were not previously recorded at floodplain sites over the last nine years of TLM condition monitoring. Key: FG = functional group: Arp = amphibious floating plants, Ate = amphibious monocotyledons, S = submerged species, Tda = terrestrial species preferring damp habitats, Tdr = drought-tolerant species; CS = conservation status in Victoria: v = vulnerable; BB = Black Box understorey community; RRG = River Red Gum understorey community.

FG	Scientific name	Common name	CS	Location (site)	Last inundated	Abun. (out of 2010)
S	*Vallisneria australis*	Eel Grass		RRG (H5A)	2015	3
Arp	*Ammannia multiflora*	Jerry-jerry	v	RRG (H4A) RRG (H6A)	2014 2015	29
Arp	*Limosella australis*	Austral Mudwort		RRG (H5A) RRG (H6A)	2015 2015	12
Arp	*Myriophyllum verrucosum*	Red Water-milfoil		RRG (H5A) RRG (H6A)	2015 2015	70
Ate	*Cyperus difformis*	Dirty Dora		RRG (H4A)	2014	2
Ate	*Eleocharis pusilla*	Small Spike-sedge		RRG (H4A)	2014	5
Tda	*Phyla nodiflora* var. *minor**			RRG (H4A)	2014	1
Tda	*Rumex crystallinus*	Glistening Dock	v	RRG (H3B)	2010–11	2
Tdr	*Actinobole uliginosum*	Flannel Cudweed		BB (H1C)	~1993–94	4
Tdr	*Eriochiton sclerolaenoides*			BB (H5C)	2014 (partial)	9

Figure 34. Abundant field of Jerry-jerry on the floodplain at the edge of Lake Lockie. Jerry-jerry is a small water-responsive herb that is listed as vulnerable in Victoria (F Freestone, December 2015).

Species richness was assessed for both River Red Gum and Black Box understorey communities. A total of 86 plant species were recorded in River Red Gum understorey sites and 62 plant species were recorded in Black Box understorey sites in 2015–16 (Figure 35).

River Red Gum understorey sites



Black Box understorey sites

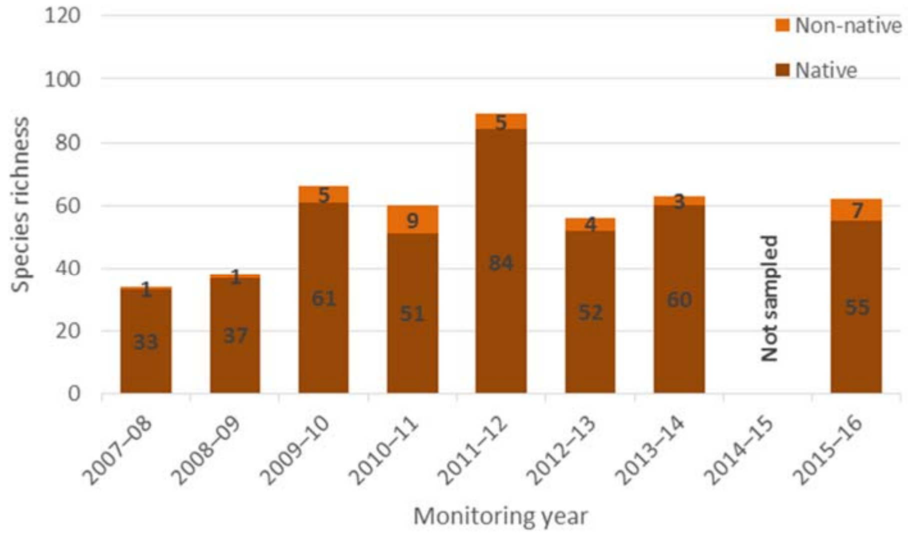
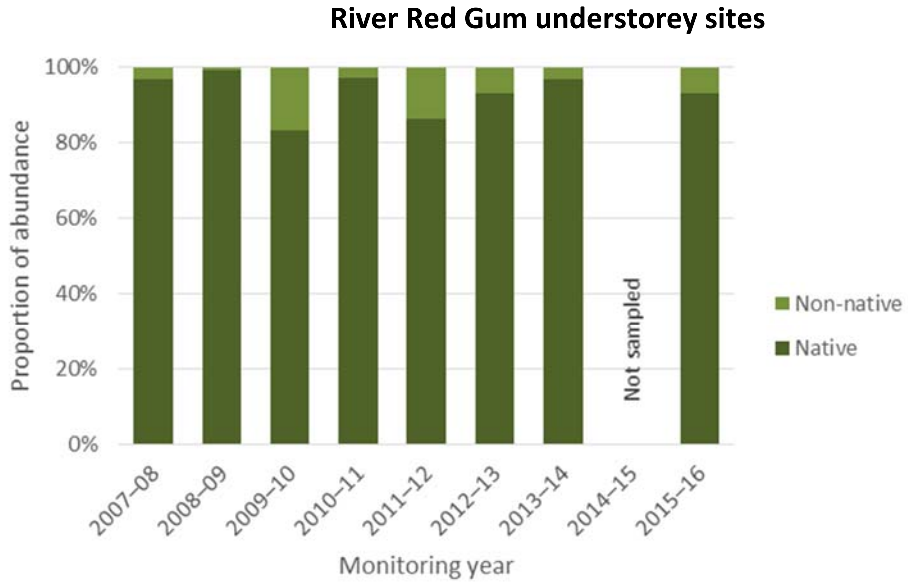


Figure 35. Number of native and non-native plant species recorded in River Red Gum and Black Box understorey sites at Hattah Lakes in all monitoring years. Not all sites were surveyed in 2010–11 due to flooding and access issues (River Red Gum: n = 11 in all years except 2010–11 where n = 9; Black Box: n = 6 in all years except in 2010–11 where n = 5).

5.4.2 Non-native species

As different numbers of sites were surveyed in different years, non-native species were assessed as a proportion of species abundance (Figure 36). Non-native species were less than 20% of the total abundance in River Red Gum vegetation communities and less than 10% of the total abundance in Black Box vegetation communities in each monitoring year.



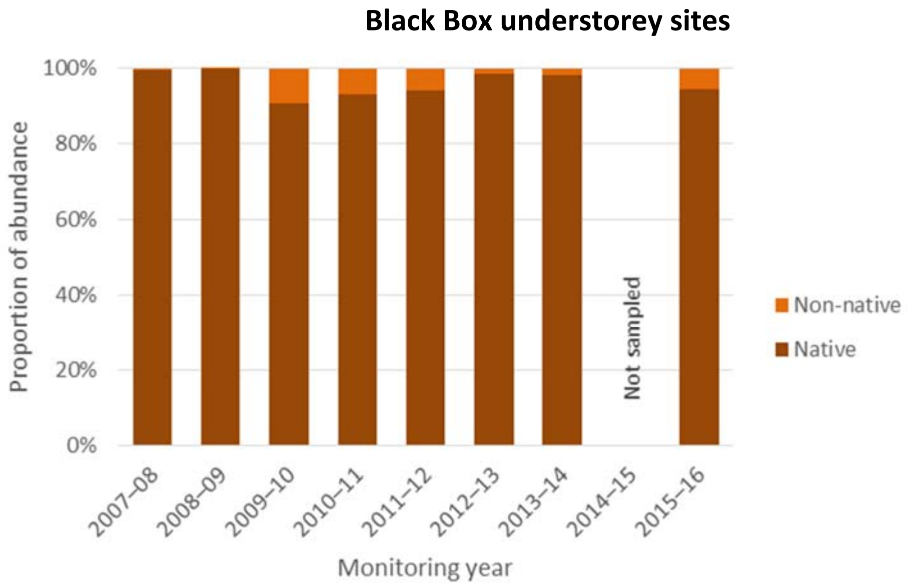


Figure 36. Proportion of non-native species abundance in River Red Gum and Black Box understorey communities at Hattah Lakes in all monitoring years (River Red Gum: n = 11 in all years except 2010–11 where n = 9; Black Box: n = 6 in all years except in 2010–11 where n = 5).

5.4.3 Rare or threatened species

Of the 113 species recorded in 2015–16, 8 are listed as having conservation significance in Victoria (Table 5.6). Surveys undertaken for TLM do not specifically target species with conservation significance. The abundance count (out of a possible 1020, being the number of 1 m x 1 m cells surveyed on the Hattah floodplain) for each species is included in Table 5.6. For comparative purposes, the most abundant species recorded at the Hattah Lakes floodplain sites in 2015–16 was Ruby Saltbush (*Enchylaena tomentosa* var. *tomentosa*), which had an abundance count of 194 (out of a possible 1020). Information on rare or threatened species recorded at floodplain sites in previous monitoring years can be found in section 4 of Part B of this report.

Table 5.6. Rare or threatened plant species recorded on the Hattah Lakes floodplain in 2015–16. Key: FG = functional group: Arp = amphibious floating plants, Tda = terrestrial species that typically occur in damp habitats, Tdr = terrestrial species that typically occur in dry habitats; CS = conservation status in Victoria: e = endangered, FFG = *Flora and Fauna Guarantee Act 1988*, k = poorly known, r = rare, v = vulnerable; WRC = water regime class: RGF = Red Gum forest, FRGW = fringing Red Gum woodland, RGFTU = Red Gum with flood-tolerant understorey, BBW = Black Box woodland. ^Habitat preference is based on Cunningham et al. (1992) and Harden (1992, 1993, 2000, 2002) and notes from previous TLM records.

FG	Scientific name	Common name	Family	CS	WRC	Habitat preference^	Abundance (out of 1020)
Tda	*Alternanthera sp. 1*	Plains Joyweed	Amaranthaceae	k	RGF	Occurs mainly on clay soils of the Riverina. Recorded in Black Box and River Red Gum understorey communities since 2010–11.	3
Arp	*Ammannia multiflora*	Jerry-jerry	Lythraceae	v	FRGW RGFTU	Grows in wet or damp conditions, often in shallow water of swamps and on river bank areas with heavy clay soils. Recorded for the first time at Hattah during TLM floodplain surveys in 2015–16.	29
Tdr	*Boerhavia coccinea*	Tah-vine	Nyctaginaceae	r	BBW	Can occur on a wide variety of soils. Recorded in Black Box woodland near the edge of Lake Lockie in 2011–12 and again in 2015–16.	4
Tdr	*Calotis cuneifolia*	Blue Burr-daisy	Asteraceae	r	RGF	Grows mostly in sandy and red clay loam soils, in a wide range of plant communities. Recorded at seven sites in 2007–08, but then not recorded again until 2011–12. In 2015–16 this species was recorded at only one site.	1
Tda	*Phyllanthus lacunarius*	Lagoon Spurge	Phyllanthaceae	v	BBW RGFTU	Occurs on most soil types in creek beds, on banks and on floodplains. Recorded in 2009–10, 2012–13 and 2015–16.	2
Tda	*Rumex crystallinus*	Glistening Dock	Polygonaceae	v	RGF	Generally grows in damp or low-lying areas, often around the perimeter of ephemeral lakes and swamps, on soil from which water	2

FG	Scientific name	Common name	Family	CS	WRC	Habitat preference^	Abundance (out of 1020)
						has receded. Recorded for the first time at Hattah floodplain sites in 2015–16 during TLM surveys.	
Tdr	*Sclerolaena muricata* var. *muricata*	Black Roly-poly	Chenopodiaceae	k	RGFTU	Widespread colonising species in NSW, especially on overgrazed or overstocked areas on heavy soils. This species is listed as poorly known in Victoria. This species has been recorded in each survey year except the very first (2007–08).	7
Tdr	*Sida ammophila*	Sand Sida	Malvaceae	v	BBW	Occurs in sandy based shores, sand hills and sandy ridges. This species has previously been recorded in BBW in most survey years.	3

5.4.4 Functional groups

Functional group abundance data is displayed for four monitoring years for both River Red Gum (Figure 37) and Black Box (Figure 38) understorey communities. Monitoring years were chosen to display changes in community composition over time, in relation to inundation disturbance.

The first year of monitoring occurred in 2007–08 during the millennium drought. Understorey composition in River Red Gum (Figure 37 (a)), and particularly in Black Box communities (Figure 38 (a)), was dominated by drought-tolerant species (i.e. species from the Tdr functional group).

In 2010–11, a natural flood inundated all of the River Red Gum and two of the Black Box understorey sites. The greatest abundance (and number of species) was recorded in 2011–12 in both communities (Figure 37(b) and Figure 38 (b)) after floodwater had receded from the floodplain. Species from an array of amphibious and terrestrial functional groups were recorded in River Red Gum understorey communities. While Black Box understorey communities remained dominated by drought-tolerant species, abundance increased dramatically in that functional group (Tdr).

The delivery of environmental water affected vegetation communities between 2013–14 and 2015–16. Environmental water inundated four River Red Gum and two Black Box sites during the 2013–14 monitoring year. In 2014, these sites were inundated again when water was delivered for a second time. One River Red Gum site (H5A at the edge of Lake Arawak) received environmental water for a third time in 2015 and remained inundated during 2015–16 monitoring.

The effect of environmental water on community composition in River Red Gum understorey sites is shown in Figure 37 (c) and (d). Species from an array of functional groups were recorded in both monitoring years, including one submerged species (functional group S) in 2015–16: Eel Grass, which is the first time a submerged species has been recorded on the floodplain at Hattah Lakes during TLM condition monitoring.

Black Box understorey communities remained heavily dominated by drought-tolerant species (Tdr functional group) in 2013–14 (Figure 38 (c)) and 2015–16 (Figure 38(d)).

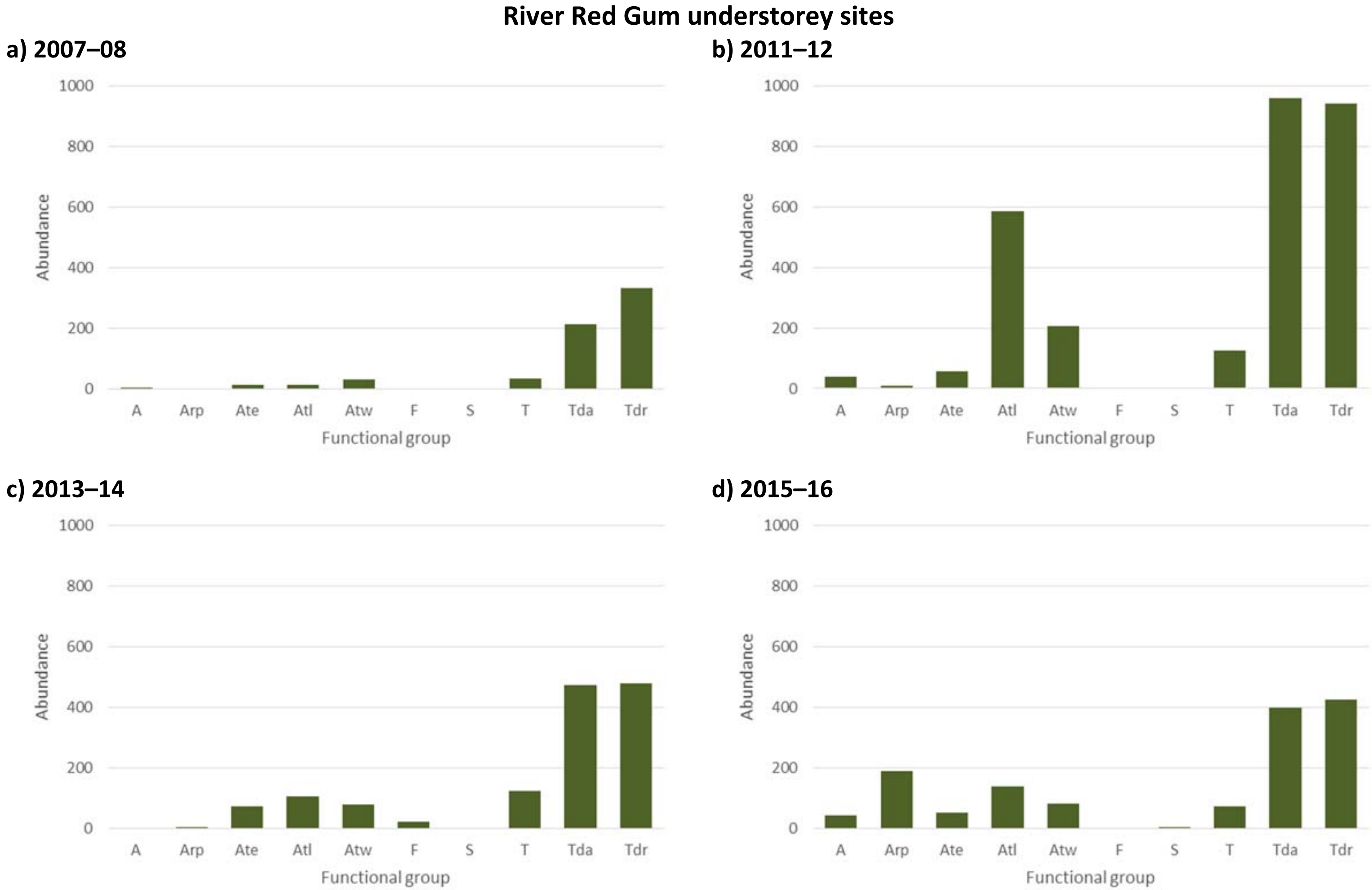


Figure 37. Functional group abundance data for River Red Gum understorey sites over four monitoring years at Hattah Lakes (*n* = 11).

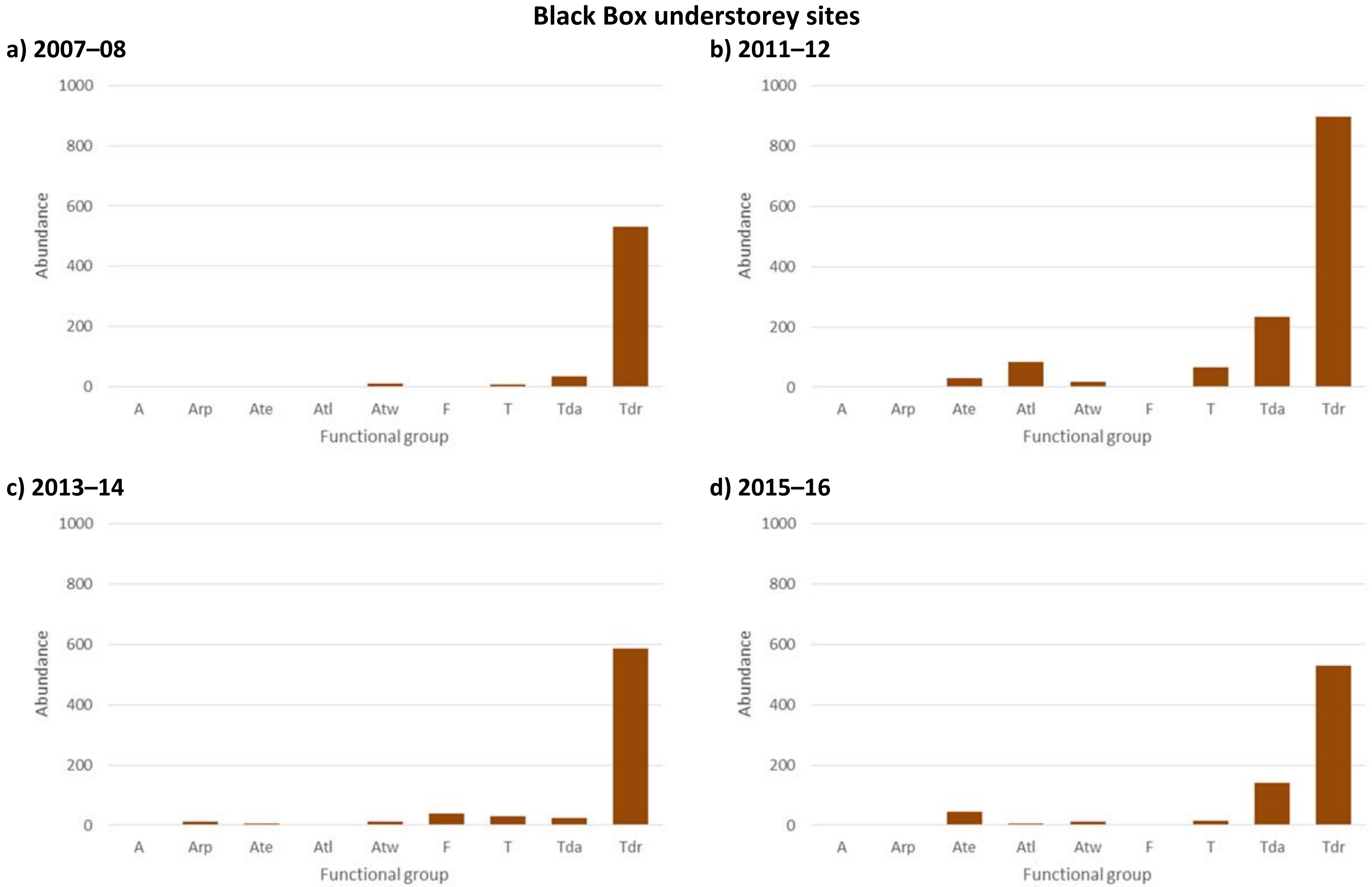


Figure 38. Functional group abundance data for Black Box understorey sites over four monitoring years at Hattah Lakes (*n* = 6).

5.4.5 Community composition based on functional groups

Community composition differed between vegetation community type and monitoring years. Because there was a significant interaction between year and vegetation community type (i.e. River Red Gum or Black Box understorey) (*P* = 0.0001), PERMANOVA pairwise tests were undertaken, based on mean functional group composition and abundance data per site, per year, for each vegetation community (Table 5.7). A statistically significant difference was found in 2015–16 compared with every other year in both understorey communities, with one exception – there was no statistically significant difference in the Black Box understorey communities between 2015–16 and 2012–13.

Table 5.7. The PERMANOVA (pairwise) test results showing differences in functional group composition and/or abundance between years, for each understorey community. Significance has been reported using Monte Carlo *P*-values and statistically significant effects are indicated in bold (α ≤ 0.05).

Community	2015–16 vs 2007–08	2015–16 vs 2008–09	2015–16 vs 2009–10	2015–16 vs 2010–11	2015–16 vs 2011–12	2015–16 vs 2012–13	2015–16 vs 2013–14
River Red Gum understorey	**0.0001**	**0.0001**	**0.0001**	**0.0001**	**0.0001**	**0.0026**	**0.0041**
Black Box understorey	**0.0069**	**0.0131**	**0.0025**	**0.0133**	**0.0323**	0.4476	**0.012**

The MDS ordination in Figure 39 displays changes in functional group composition over time. Mean composition and abundance data per vegetation community for each year was used for display purposes. Drought years (2007–08 to 2009–10) in both communities were heavily dominated by species from the terrestrial dry (Tdr) functional group. River Red Gum understorey sites in 2010–11 were dominated by species from the floating (F) functional group, as all sites were inundated during that monitoring year. In the years since the natural flood (i.e. 2011–12 to 2015–16), River Red Gum understorey sites recorded species from an array of amphibious and terrestrial functional groups, initially in response to flooding and then maintained at a number of sites by environmental water in more recent years. Black Box understorey communities remain dominated by drought-tolerant species (Tdr functional group).

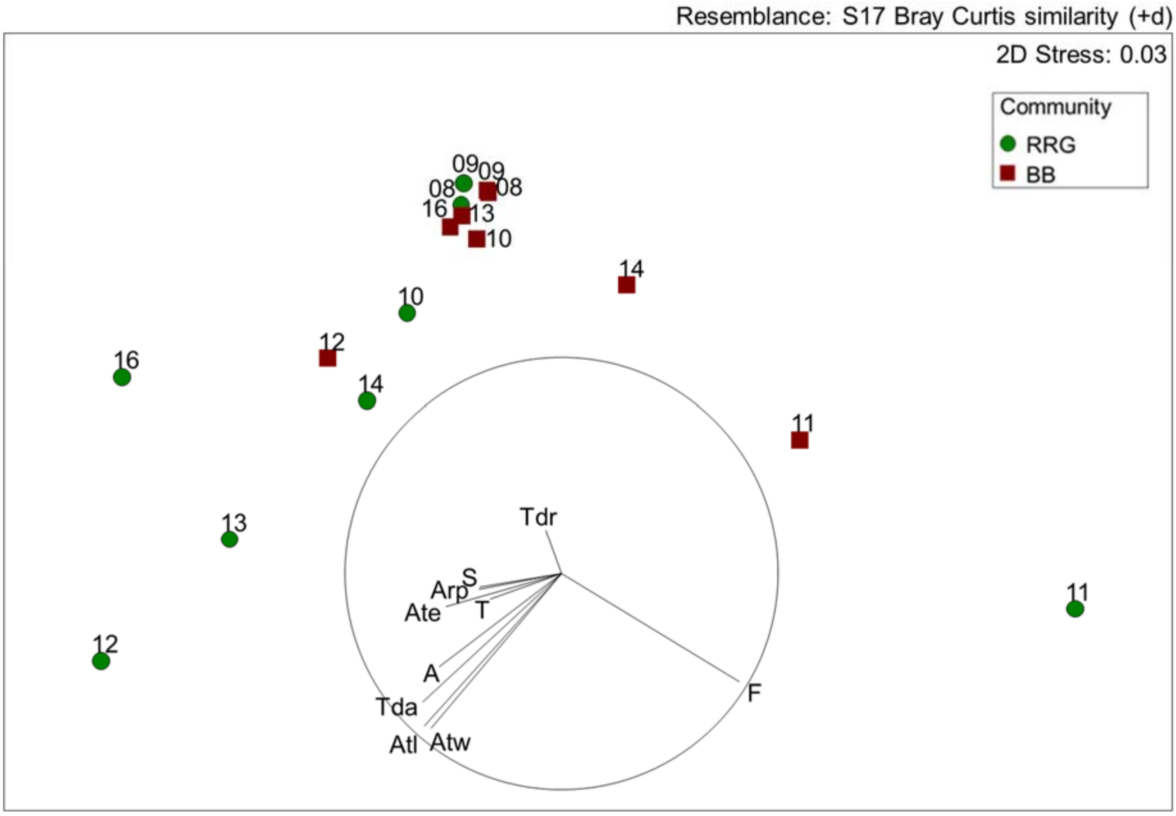


Figure 39. Differences in floodplain vegetation functional group composition at Hattah Lakes between vegetation communities and monitoring years. Key: BB = Black Box understorey community, RRG = River Red Gum understorey community.

5.5 Discussion

Sections of the water-responsive plant community on the Hattah Lakes floodplain have benefited from a return to favourable environmental conditions since 2010. Ten species recorded in 2015–16 had not been previously recorded over the last nine years of TLM floodplain condition monitoring. Eight out of these ten were recorded at locations that were inundated by natural flooding in 2010–11, followed by delivery of environmental water on multiple occasions between 2013 and 2015. The majority of these species (i.e. seven out of eight), are considered water-responsive plants (e.g. from the submerged, amphibious or terrestrial damp functional groups). This is also the first time a species from the submerged functional group, Eel Grass, was recorded on the Hattah floodplain during TLM condition monitoring. Although new to the dataset for TLM floodplain condition monitoring, Jerry-jerry, listed as vulnerable in Victoria, and Red Water-milfoil, which was abundant in Hattah Lakes wetlands in 2015–16, had been previously recorded in 2014 during TLM intervention monitoring (Freestone et al. 2014). It was also during this intervention monitoring that four other amphibious species were recorded for the first time at TLM floodplain sites (Waterwort, Narrow-leaf Nardoo *Marsilea costulifera*, Pale Knotweed *Persicaria lapathifolia* and Furrowed Pondweed). The appearance of these species on the floodplain indicates that the water-responsive plant community, particularly in River Red Gum understorey communities, has benefited from inundation.

5.5.1 Species richness and abundance

A total of 113 species were recorded across all floodplain sites at Hattah Lakes, including eight species listed as having conservation significance in Victoria. Jerry-jerry and Glistening Dock, both listed as vulnerable in Victoria, were recorded for the first time in 2015–16 under TLM condition monitoring of floodplain sites. Though new to this dataset, Jerry-jerry has previously been recorded at Hattah Lakes (EPA & MDFRC 2008; Freestone et al. 2014; Henderson et al. 2012) and Glistening Dock was recorded at two wetland sites in 2015–16 (see section 4.4.1 and 4.4.3).

Species richness increased from the previous monitoring year (2013–14) and in 2015–16 was the second highest over nine years of condition monitoring. This is likely a reflection of water recession from the floodplain following repeated inundation at 6 of the 17 sites over the last few years. Floodplains support a high diversity of ephemeral plants that only appear following inundation (Brock et al. 2006). The greatest number of understorey species recorded was in 2011–12, following natural flooding and above-average rainfall. In 2015–16, the four most common species recorded included two from the drought-tolerant functional group (Tdr) and two from water-responsive functional groups (Atl and Tda).

For riverine grassy woodlands or forests in Victoria, the average relative non-native species cover is 20–25% and relative non-native species richness is 30–35% (Catford & Jansson 2014). In all years, relative non-native species richness and abundance was typically less (or in line with) the state average for these vegetation community types.

5.5.2 Community composition based on functional groups

The use of functional groups is a widely accepted method of interpreting changes in plant communities in relation to disturbance, while minimising the effects of changes in species composition or inconsistencies in taxonomic classification (Brock & Casanova 1997; Campbell et al. 2014). Functional groups demonstrate the impact of flood inundation on community composition. In drought years, understorey composition in both River Red Gum and Black Box communities were dominated by species from the terrestrial dry (Tdr) functional group. In 2009–10, species richness in both River Red Gum and Black Box understorey communities increased due to rainfall events that occurred prior to the survey. Although there was a substantial increase in species richness, the community composition remained largely dominated by Tdr species, demonstrating the influence of rainfall in these semi-arid environments. The greatest change in community composition occurred following the natural flood (2011–12). As flood water receded from the floodplain, species from an array of terrestrial and amphibious functional groups were recorded in both River Red Gum and Black Box understorey communities that were inundated. Due to the extent of inundation, the greatest change in functional group composition was observed at River Red Gum sites. This change in functional group composition demonstrates the importance of flood inundation to floodplain understorey vegetation communities, compared with rainfall alone.

In 2015–16, species from an array of functional groups continued to be recorded at River Red Gum understorey communities. Between 2013–14 and 2015–16 there has been an increase in the abundance of amphibious floating (Arp) species and the first records of submerged (S) species at River Red Gum understorey communities as a result of both the delivery and recession of environmental water at four of eleven sites on multiple occasions since flooding in 2010–11. This demonstrates the ability of environmental water to maintain aquatic and floodplain plant communities at these sites since flooding in 2010–11.

In contrast, Black Box understorey communities remain dominated by drought-tolerant species (functional group Tdr). Interestingly, Black Box community composition recorded in 2015–16 is most similar to that recorded in 2012–13. It is possible that similarities are because of the similar length of time since inundation at two of the six sites (e.g. inundation via natural flood in 2010–11 occurred two years prior to 2012–13 surveys and inundation in spring 2014 via environmental flows occurred one year prior to 2015–16 surveys). It would be beneficial to analyse data from these two surveys to investigate similarities between natural flooding and environmental flows.

Even though two of the six Black Box sites were inundated by environmental water in 2014, four sites (and a large portion of the Hattah Black Box understorey community) have not been inundated by overbank flows since ~1993–94. Above-average rainfall received in the summer of 2010–11 likely brought some relief to these communities after the millennium drought, however these sites remain dominated by drought-tolerant plants. The ideal flooding frequency for River Red Gum communities is once every three to five years and Black Box woodland communities at least once every 10 years (Rogers & Ralph 2011). While an ideal flooding frequency has been achieved for a proportion of River Red Gum communities at Hattah Lakes in recent years (2010–11 flood and environmental water in 2013, 2014 and 2015), a large portion of the Black Box understorey communities are in need of flood inundation.

5.5.3 Recommendations

To improve our understanding of the benefits of multiple flows on floodplain vegetation, it would be beneficial to analyse species abundance and diversity data in relation to inundation history. This includes analysing vegetation response to inundation at sites that have received multiple flows (2010–15) compared to sites that are long dry (River Red Gum last inundated 2010–11; Black Box last inundated 1993–94). This could be undertaken using data that has already been collected through TLM condition and intervention monitoring.

Consecutive environmental flows appear to be supporting flow-dependent rare plants. As discussed in section 4.5, targeted surveys (timed to coincide with the recession of water from floodplains following the delivery of environmental water) are recommended and would provide valuable information for the management of flow-dependent rare plants.

5.5.4 Summary

Key points from condition monitoring of floodplain understorey vegetation at Hattah Lakes in 2015–16 are:

- Water-responsive species have benefited from a return to favourable environmental conditions since 2010, particularly in River Red Gum communities. This is likely in response to flows inundating a portion of the floodplain on multiple occasions over the last few years (e.g. flooding in 2010–11 followed by delivery of environmental water in 2013, 2014 and 2015).
- Ten species were recorded for the first time at Hattah floodplain sites in the last nine years of TLM condition monitoring (Figure 40 (Left)). Two of these species are listed as vulnerable in Victoria: Jerry-jerry and Glistening Dock (Figure 40 (Right)).
- Most Black Box understorey vegetation communities have not been inundated for an extended period of time (~20 years). Ideal flooding frequency for Black Box woodland communities is inundation once in every 10 years (Rogers & Ralph 2011).

Figure 40. A carpet of Red Water-milfoil at the edge of Lake Lockie) (Left). This species was recorded at floodplain sites for the first time in 2015–16 (F Freestone, December 2015). Jerry-jerry at Hattah Lakes (Right). This species was recorded at floodplain sites for the first time in 2015–16 (F Freestone, December 2015).

6 Lignum

FIONA FREESTONE

6.1 Introduction

Tangled Lignum (*Duma florulenta* (Meisn.) T. M. Schust; formerly known as *Muehlenbeckia florulenta* Meisn.), hereafter referred to as Lignum, is considered one of the most ecologically significant floodplain shrubs of Australia (Roberts & Marston 2011). Lignum is a native shrub that can form dense thickets, dominating large areas of floodplain throughout the Murray–Darling Basin (Cunningham et al. 1992; Sainty & Jacobs 1981). Lignum provides primary habitat for birds, reptiles and mammals and is significant as breeding habitat for many colonially nesting waterbirds (Maher & Braithwaite 1992). During inundation, the structure of Lignum provides shelter for fish and aquatic invertebrates (Roberts & Marston 2011) and during dry periods Lignum facilitates the growth of floodplain understorey herbs (James et al. 2015).

Lignum condition is strongly influenced by soil moisture and is therefore highly dependent on flood regimes in arid areas, where rainfall alone is unlikely to sustain these communities (Craig et al. 1991). River regulation has led to a reduction in the frequency, magnitude and duration of flooding in the lower reaches of the Murray–Darling system (Leblanc et al. 2012). It has been suggested that Lignum requires flooding every 3 to 10 years for periods of up to 12 months, and is intolerant to both sustained dry periods and prolonged flooding (Craig et al. 1991). The Living Murray program is a large-scale restoration project that attempts to ameliorate the negative effects of regulation on wetlands and the floodplain. Condition monitoring of Lignum at the Hattah Lakes icon site has been undertaken annually since 2006–07. This chapter reports on changes in Lignum condition between 2006–07 and 2015–16.

6.2 Ecological objectives

The vision for the Hattah Lakes icon site is to:

> *Preserve and, where possible, enhance the biodiversity values of the Hattah Lakes; and restore healthy examples of all original wetland and floodplain communities that represent the communities which would be expected under natural flow conditions (MDBA 2012c).*

The aim of this chapter is to report on the change in Lignum condition over time at Hattah Lakes. The Living Murray program is currently undergoing refinements that involve the development of operational objectives and identification of suitable ecological indices that link back to the vision statement. As part of these refinements, recommendations include changes to the methods of collecting Lignum data. At the time this report was compiled, new methodology for data collection had not yet been adopted. As an interim measure for this monitoring period, this chapter reports on the established vision statement by examining changes in Lignum condition over time. This will be achieved by analysing changes in colour and viability of individual plants that have been surveyed repeatedly over the last 10 years of condition monitoring.

6.3 Methods

Lignum condition was assessed using two methods:

1. Condition over time: 10 years of data were analysed to identify trends over time.
2. Three survey comparison: three survey years were analysed using a larger data set as a better representative sample of the icon site (i.e. including additional sites established in 2012–13).

6.3.1 Sites

There are 10 established Lignum monitoring sites at Hattah Lakes (H1 to H10), each containing 30 tagged Lignum plants. Sites were selected to represent the various EVCs, WRCs and elevations of the floodplain. Sites H1 to H5 were established in 2006–07 and were reassessed in summer 2007–08 and 2008–09 and then annually in spring for subsequent years (excluding 2014–15 when no data was collected due to changes in funding). An additional five sites were established in 2012–13 to provide a more representative sample of the Hattah Lakes icon site (H6 to H10). To compensate for the loss of plants through mortality (due to extended drought) or missing plants (plants that were not able to be located due to missing tags and recorded as no data), an additional 26 plants were added to the survey in 2012–13. These plants were located as geographically close to the original plants as possible and were monitored in 2012–13, 2013–14 and 2015–16 and will continue to be surveyed annually in spring. Location details, maps and photos taken at each site in each survey year are provided in section 5 in Part B of this report.

6.3.2 Hydrology

Site hydrology was determined in order to provide context to condition monitoring data. Flooding history for each site was estimated using RiM-FIM (Overton et al. 2006) and Murray River flows recorded downstream of Euston Weir (data courtesy of MDBA). Natural flooding of the Murray River, Hattah Lakes and its associated floodplains beginning in spring 2010 resulted in the inundation of five Lignum sites at Hattah Lakes (Table 6.1). Based on RiM-FIM and digital elevation modelling (DEM), the remaining sites were most likely last inundated in 1993–94 or earlier and have received only rainfall since. As there are spatial limitations associated with using RiM-FIM, in some instances site observations were used where inaccuracies were evident in GIS data.

Site H1, adjacent to Lake Lockie, received environmental water in 2005, 2006, 2009, 2010, 2013, 2014 and 2015 (Table 6.1). Environmental water delivered in 2014 inundated H6 on Moonah Track and H9 at the edge of Lake Yerang. Sites H2 near Lake Brockie and H7 between Chalka Creek (south) and Lake Cantala, also appeared to have been at the fringe of inundation in 2014. All sites were dry during surveys in 2015–16 with the exception of H1 at the edge of Lake Lockie, which was inundated to approximately 30 cm.

Table 6.1. Hydrology for TLM Lignum condition monitoring sites at Hattah Lakes.

Site	Last natural flood		Environmental water delivered
	1993	2011	
H1		✓	2005, 2006, 2009, 2010, 2013, 2014 and 2015
H2		✓*	2014
H3	✓		
H4	✓^		
H5	✓		
H6		✓*	2014
H7		✓	2014
H8	✓^		
H9		✓*	2014
H10	✓		

*Observations confirm that this site was inundated in 2011, although RiM-FIM data disagrees.
^RiM-FIM data not available at these sites. Flow level obtained using DEM (spatial layer provided by the Mallee Catchment Management Authority) and compared with sites at similar elevations.

6.3.3 Analysis

The Lignum Condition Index (LCI) method used to assess the condition of Lignum at Hattah Lakes is outlined in section 7 of The Living Murray: Condition Monitoring Program design for Hattah Lakes (MDFRC 2011). The LCI is the sum of the scores for two plant condition indicators: viability and colour (Table 6.2). In some survey years, plants were not able to be located, either due to missing tags or inundation. Plants that were not able to be located and assessed were recorded as no data.

Table 6.2. Viability and colour scores used to assess the condition of Lignum plant condition. Adapted from Scholz et al. (2007b).

% Viable	Score
> 95	6
75 ≤ 95	5
50 ≤ 75	4
25 ≤ 50	3
5 ≤ 25	2
0 ≤ 5	1
0	0

Colour	Score
all green	5
mainly green	4
half green, half yellow/brown	3
mainly yellow/brown	2
all yellow/brown	1
no viable stems	0

Condition over time

To evaluate change in Lignum condition over time, scores for each of the two LCI indicators for each plant were added together and then categorised into one of five condition categories (Table 6.3). Change in Lignum condition over 10 years was assessed using data from sites established in 2006–07 (i.e. H1 to H5). One-way analysis of variance (ANOVA) was used to determine if there were statistically significant differences in mean LCI between years. Holm-Sidak pairwise comparisons (SigmaPlot version 11; Systat Software, San Jose, CA) were then undertaken to determine which pairs of years were significantly different.

Table 6.3. Condition categories for LCI combined scores.

LCI combined score	Condition category
0	zero*
1 to 3	poor
4 to 6	moderate
7 to 9	good
10 to 11	very good

*The category zero relates to Lignum plants that were observed to have no viable above-ground biomass. These plants were presumed to be either dead, or dormant and persisting as viable underground rootstock.

Three-survey comparison

Three years of data were compared for plants surveyed in 2012–13, 2013–14 and 2015–16 using the condition categories described above (Table 6.3). This analysis excluded plants that were missing or considered dead with no sign of regeneration following the natural flood in 2010–11 (i.e. it included all plants that were recorded as alive in 2012–13). A one-way analysis of variance (ANOVA) was used to determine if there was a statistically significant difference in mean LCI between the three surveys (SigmaPlot version 11; Systat Software, San Jose, CA).

6.4 Results

6.4.1 Condition over time

Condition results for plants that were assessed annually from 2006–07 to 2015–16 (excluding 2014–15) are presented in Figure 41. The highest proportion of plants with an LCI score of zero occurred in 2010–11. The decrease in the number of plants recorded as zero after 2010–11 indicates regeneration from surviving rootstock following natural flooding. The greatest number of plants recorded in good condition occurred in 2012–13, though no plants have been recorded in the very good category since 2009–10. In 2015–16, the number of plants recorded in good and moderate condition had decreased since 2013–14 and the number of plants recorded in poor condition had increased.

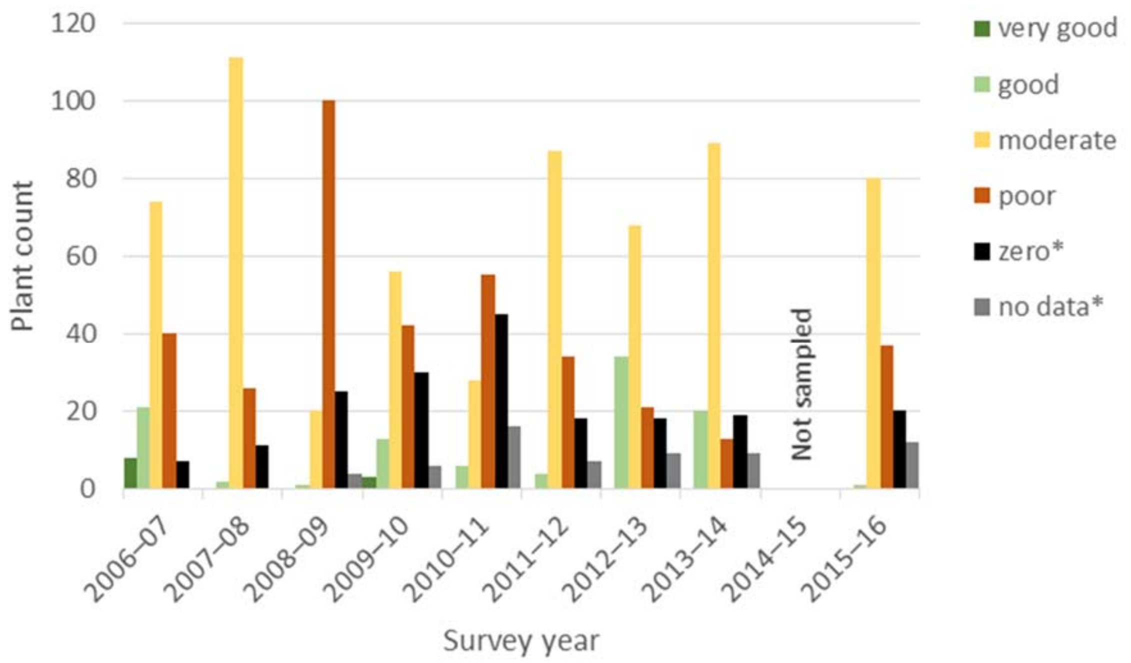


Figure 41. Count of plants in each LCI condition category across the Hattah Lakes icon site for each survey year (sites H1 to H5; *n* = 150 plants per year).

Key: *the category zero relates to Lignum plants that were observed to have no viable above-ground biomass. These plants were presumed to be either dead, or dormant and persisting as viable underground rootstock. Note: in 2012–13, 26 Lignum plants were recorded as zero (n = 18) or no data (n = 8) for two or more years and showed no sign of regeneration following the natural flood (summer of 2010–11). These plants were not specifically assessed in 2013–14 and 2015–16 as they showed no sign of recovery following favourable environmental conditions. For the purposes of this graph, data for these plants have been extrapolated from 2012–13 records in 2013–14 and 2015–16 (i.e. 18 zero records and 8 no data records in 2012–13 were added to the 2013–14 and 2015–16 data, though not explicitly surveyed).

The LCI scores were pooled per survey to determine average Lignum condition for each year. Figure 42 shows the change in average Lignum condition per year from 2006–07 to 2015–16 (excluding 2014–15). The average condition has declined since 2012–13.

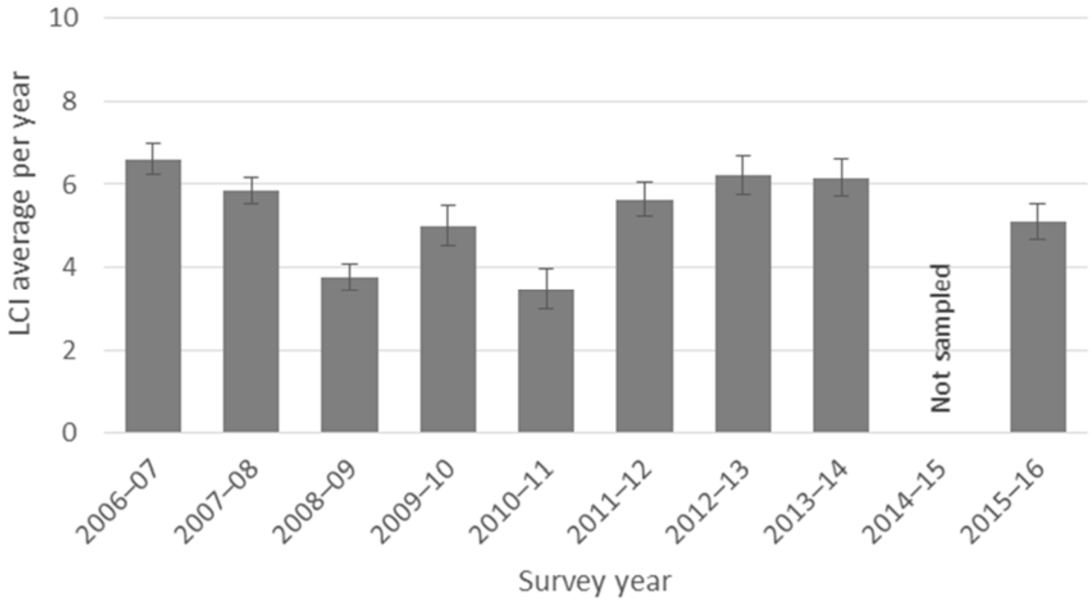


Figure 42. Average (± 95% CI) LCI for each survey at the Hattah Lakes icon site (sites H1 to H5; *n* = 150 plants per year).

One-way analysis of variance using mean LCI scores identified a significant difference among survey years ($P < 0.001$). Holm-Sidak pairwise comparisons showed statistically significant differences between some, but not all, years. Pairwise comparisons for the current survey year (2015–16) and all other survey years are presented in Table 6.4. There was a statistically significant difference between 2015–16 survey data and the previous two surveys following a decline in Lignum condition.

Table 6.4. Holm-Sidak pairwise comparisons showing statistically significant differences (bold) between LCI Scores in 2015–16 and all other survey years at Hattah Lakes.

Survey years	**2006–07**	**2007–08**	**2008–09**	**2009–10**	**2010–11**	**2011–12**	**2012–13**	**2013–14**
2015–16	**<0.001**	0.140	**<0.001**	0.943	**<0.001**	0.510	**0.003**	**0.007**

6.4.2 Three-survey comparison

The condition of plants at all sites (H1 to H10, including all replacement plants and excluding dead or missing plants that were discontinued in 2012–13) were compared between survey years 2012–13, 2013–14 and 2015–16 (Figure 43). In each survey, the majority of plants were recorded as being in moderate condition. Only two plants were recorded as being in very good condition in 2012–13. There is a general shift from the number of plants recorded in good condition to the number of plants recorded in moderate and poor condition in 2015–16.

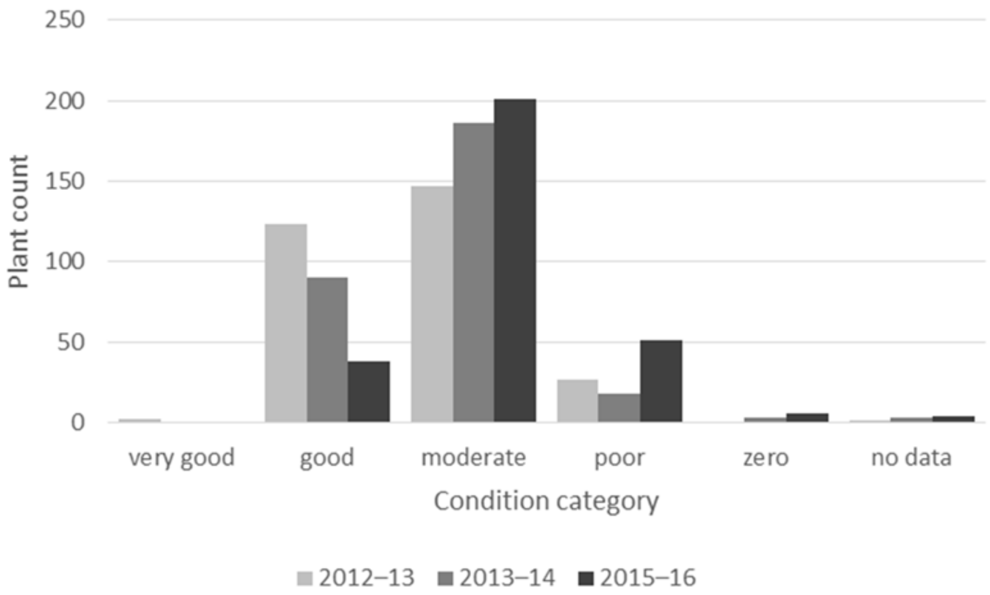


Figure 43. Count of LCI condition categories of all sites (including replacement plants and excluding discontinued plants) at Hattah Lakes (*n* = 300 plants per survey).

The mean LCI score was 7.77 in 2012–13, 7.57 in 2013–14 and 6.77 in 2015–16. This shift was associated with a decrease in the number of plants in good condition and an increase in the number of plants in moderate and poor condition in 2015–16. Using ANOVA, 2015–16 was statistically different to both 2012–13 (P = <0.001) and 2013–14 (P = <0.001) however, no statistically significant difference was found between 2012–13 and 2013–14 (P = 0.126).

6.5 Discussion

Over 10 years of TLM condition monitoring of Lignum, Hattah Lakes has experienced severe drought (mid-1990s to 2009), followed by widespread flooding (late 2010–11) (Leblanc et al. 2012). Higher-than-average rainfall in late 2010–11 across the Murray River catchment caused overbank flooding of the Murray River into Hattah Lakes and the surrounding floodplain. Areas of floodplain that were not inundated by overbank flooding received substantial rainfall during the summer of 2010–11 (573.6 mm, compared with a long-term average of 167.8 mm, at Ouyen) (BOM 2016). Environmental flows inundated a portion of the Lignum community in 2014 and 2015.

Lignum communities appear to have benefited from flooding and/or rainfall since 2010–11, as indicated by the increased average LCI condition scores for three years following the flood and/or rainfall. LCI condition categories show that the number of plants recorded as zero (having no viable above ground biomass) was highest during monitoring in spring of 2010–11, towards the end of the extended drought (monitoring occurred prior to the 2010–11 flood). Lignum can withstand drought in a dormant state, regenerating from viable underground rootstock when environmental conditions become favourable again (Brock et al. 2006; Roberts & Marston 2011). More than half of the plants recorded as zero during the last year of drought (2010–11) regenerated following the natural flood (2011–12 survey). It is likely that these plants regenerated from viable underground rootstock. An increase in soil moisture as a result of flooding and/or rainfall enables a rapid growth response in Lignum while this water source is readily available (Chong & Walker 2005; Jensen 2008). This rapid response is reflected in the improvement of Lignum condition in surveys following natural flooding (e.g. 2011–12 and 2012–13).

Following recession of the natural flood, as the floodplain continued to dry, Lignum condition began to decline from 2012–13. Condition in 2015–16 more closely resembles that recorded in 2009–10. The statistically significant difference between Lignum condition in 2015–16 and the previous two surveys (2012–13 and 2013–14) is evidence of this decline, even though the average Lignum condition remains in the moderate condition category. This result is contradictory to expectations. Half of the sites surveyed in 2015–16 received environmental water during 2014 and it was anticipated that there would be an improvement in condition between 2013–14 and 2015–16. It is possible that condition has deteriorated at sites that did not receive environmental water, counteracting the benefits of water delivery at an icon-site scale. Or, it is possible that condition did not improve at sites that did receive environmental water, though this seems unlikely as casual observations in 2015–16 indicate that plants at these sites had experienced growth spurts since previously surveyed (e.g. plant tags were hidden well within Lignum shrubs, indicating branch growth around them). An analysis between sites that were inundated by environmental water and sites that were not was outside the scope of this report, though this is recommended in order to improve understanding of Lignum condition and response to environmental watering at Hattah Lakes.

Overall, Lignum was in moderate condition at the Hattah Lakes icon site in 2015–16. The recommended flood frequency for Lignum and Lignum communities is once in every 3 to 10 years (Rogers & Ralph 2011). While dry periods may inhibit optimal Lignum growth, the known distribution of Lignum indicates that Lignum plants have the ability to respond to favourable environmental conditions after up to 10 years of inter-flood dry-periods (Rogers & Ralph 2011). The slight decline in Lignum condition over the last few survey years indicates an overall drying of the floodplain at Hattah Lakes following recession of the natural flood in 2010–11. Even though half the sites were inundated by environmental water in 2014, the other half are long-dry (i.e. not inundated by overbank flows for 20+ years) and it is possible that their condition may be strongly influencing the overall result for the Hattah Lakes icon site.

6.5.1 Recommendations

Half the sites surveyed were inundated by environmental water in 2014, while the other half are long-dry (i.e. not inundated by overbank flows for 20+ years). It is possible that condition of the long dry sites may be strongly influencing the overall result for the Hattah Lakes icon site (e.g. trend towards a decline in condition). It would be beneficial to analyse differences between these sites to improve our understanding of Lignum condition and response to environmental watering at Hattah Lakes.

Due to the sampling methods, the data recorded in 2006–07 is not necessarily representative of antecedent conditions at that time. All plants initially selected and surveyed were alive in the first survey, so any accumulated previous mortality was excluded. This is one of the main drawbacks to this method of data collection and why changes to the method, recommended in a recent review (Robinson 2014a, 2014b), will be implemented at the next opportunity.

6.5.2 Summary

Key points for Lignum condition monitoring at Hattah Lakes in 2015–16:

- Overall, Lignum at Hattah Lakes was in moderate condition in 2015–16 (Figure 44).
- Lignum condition has declined since 2012–13 as the floodplain continues to dry following the natural flooding and above-average rainfall in late 2010–11. This indicates that the effects of the flood (e.g. soil moisture on the floodplain) are diminishing across the icon site.

Figure 44. Lignum in moderate condition at Hattah Lakes in 2015–16 (Site H7, F Freestone, September 2015).

7 Fish

DAVID WOOD, PAUL BROWN AND SCOTT HUNTLEY

7.1 Introduction

Prior to the mid-1930s the Hattah Lakes experienced a greater frequency and variability of flooding than has occurred after that time. The construction of regulators and weirs and water extraction for agriculture, industry and urban use have reduced this frequency and variability (Souter 2005). To compensate for low flows, particularly during the later stages of the millennium drought (2001–2009), environmental water was delivered to the Hattah Lakes to provide refuge habitat for water-dependent species (MDBC 2005). Overbank pumping from the Murray River to periodically inundate wetland and floodplain habitats began in 2005 and continued annually (with the exception of 2007) until natural flooding occurred in 2010 (EPA & MDFRC 2008).

Off-channel habitats, such as the Hattah Lakes, are important to fish communities. They provide habitat diversity which is important for reproduction (nursery zones), feeding (high productivity) and heightened survival (complex snags) for multiple life stages of fish (Junk et al. 1989; Lyon et al. 2010; Souter 2005). Inundation of the floodplain not only triggers fish to move onto the floodplain but allows them to benefit from associated high productivity pulses (Junk et al. 1989; Junk & Wamtzem 2004; Lyon et al. 2010).

Early emergency watering to the Hattah Lakes facilitated a one-way connection from the river channel to the Hattah Lakes for fish. The inundation of Hattah Lakes via pumps created a 'filtered' fish assemblage that was predominantly void of non-native species (Ellis & Wood 2011; Vilizzi et al. 2013). While native species in the lakes benefited greatly from the highly productive wetland habitat, their passage back to the river was impeded and they perished as the wetlands dried. Natural flooding in 2010 and 2011 resulted in a homogenous fish assemblage between the river and lakes and unrestricted two-way passage. Following flooding, after the lakes had disconnected, non-native species, in particular Common carp (*Cyprinus carpio*), numerically dominated the fish community at Hattah Lakes. This is a relatively common occurrence in Murray–Darling Basin wetlands following flooding (Brown et al. 2005; Stuart & Jones 2006).

In 2013, the completion of new infrastructure enabled environmental flows to be delivered into the Hattah Lakes system and held higher up on the floodplain than had been possible with previous managed watering events. Regulatory structures on Chalka Creek allow water to be released back to the river, effectively allowing opportunities for fish passage back to the Murray River. Management of this infrastructure can also be used to cue particular fish species (e.g. Golden perch) to return to the river from the lakes (Wood & Brown 2016).

Condition monitoring reports on the change in environmental condition at the icon-site scale over time. Monitoring is specifically tailored to determine if manangement objectives are being met. Spatial and temporal variation and trends in native fish population demographics (i.e. diversity, spatial distribution and abundance) are interpreted with respect to progress towards achieving ecological objectives.

7.2 Ecological objectives and indices

Ecological objectives for Hattah Lakes have been in refinement since the interim objectives were first developed by the Murray–Darling Basin Ministerial Council in 2003 (MDBMC 2003). The revised ecological objectives for fish at Hattah Lakes incorporate an understanding of environmental responses learned through monitoring, evaluation, research, and modelling and consultation activities over the past decade (MDBA 2012c).

The overarching objective is:

> *Maintain high-quality habitat for native fish in wetlands and support successful breeding events.*

More detailed objectives are to:

> *Increase the distribution, number and recruitment of local wetland fish—including hardyhead, Australian smelt and gudgeon—by providing appropriately managed habitat.*
>
> *Maximise use of floodplain habitat for recruitment of all indigenous freshwater fish.*

An independent review into TLM icon site condition monitoring plans (Robinson 2013) provided a range of recommendations on how to better report on the ecological condition of icon sites. Emphasis was placed on a review of current objectives and how to better align them with data that has already been collected.

For this report, it is assumed that all native fish species that have been sampled in either the wetland or anabranch macrohabitats over the course of the TLM project are 'local wetland fish' as referred to in the overarching objective (above).

In the Hattah Lakes' wetlands and Chalka Creek, recruitment success is not dependent upon successful reproduction alone; for some species it requires, and for others it is enhanced by, immigration via pumped or natural flows from the River Murray. For species such as Golden perch (*Macquaria* ambigua) and Silver perch (*Bidyanus bidyanus*), riverine habitat and flow stimuli are necessary to complete the full life-cycle. Mature adults spawn in a riverine environment and juveniles with access to newly inundated wetlands have enhanced growth and survival (Baumgartner et al. 2014; King et al. 2009). While the Hattah Lakes can, and probably do, function as nursery habitat for juvenile Golden and Silver perch (Brown et al. 2015b), these species cannot truly 'recruit' until they re-gain access to the riverine environment from which they came—the Murray River. Small-bodied native fish and Bony herring (*Nematalosa erebi*) can complete their whole life-history within the wetland habitats of the Hattah Lakes. Hence, the objective *Maximise the use of floodplain habitat for recruitment of all indigenous freshwater fish* requires consideration of managing the re-connection of flooded habitats within the icon site to the Murray River. Progress towards this has been made by the Mallee Catchment Management Authority, largely under The Living Murray Intervention Monitoring program (Brown et al. 2015b; Freestone et al. 2014; Wood & Brown 2016).

In order to evaluate progress towards the above objectives, five indices were used:

- α species diversity index
    - within-macrohabitat diversity, where macrohabitat refers to flow guild (riverine, anabranch, wetland)
- β species diversity index
    - inter-macrohabitat diversity, where macrohabitat refers to flow guild (riverine, anabranch, wetland)
- extent index
    - the percentage of sites in which each species occurred

  - calculated for each macrohabitat as the average across all species
- nativeness index
  - the percent biomass of native compared to all fish
- expectedness index
  - the average observed-to-expected score for each reach/wetland within each year for each macrohabitat.

7.3 Methods

7.3.1 Sampling

Sampling has occurred annually on 10 occasions since 2005 (with the exception of 2015) (Table 7.1). The most recent survey was conducted during April 2016 (2015–16 monitoring period) and included five wetlands (excluding Lakes Lockie and Little Hattah as they were dry), the adjacent Murray River and Chalka Creek.

A nested sampling design consisting of sites, within reaches, within macrohabitats, was used to assess the condition of fish assemblages across the Hattah Lakes system (MDFRC 2011). These methods are consistent with those prescribed in TLM Fish Condition Monitoring Approach v.2 (MDBA 2010). Chalka Creek (i.e. channel macrohabitat) is sampled to determine what part of the fish community is occupying this different habitat type. The Murray River adjacent to Hattah Lakes (i.e. riverine macrohabitat), which is the source of fish in the Hattah Lakes system, is sampled as a comparison. Comprehensive details of the sampling design and methods used and sites sampled are contained in section 10 of MDFRC (2011).

In 2008–09, backpack electrofishing, seine netting and bait traps were introduced to complement small and large fyke netting in the anabranch and wetland macrohabitats. Additionally, bait traps, seine nets and small fyke nets were introduced to complement boat electrofishing in the riverine macrohabitat. In figures, pre-2009 data are represented by 'patterned' bars whereas data from 2009 and onwards are represented with 'solid' bars.

Table 7.1. Sites sampled for fish at the Hattah Lakes icon site from 2005 to 2016 within each reach/wetland. The month of each survey and the corresponding monitoring year are included. (Blue shaded cells indicate inundation; tick indicates site was sampled; solid red line indicates complete drying of all wetlands).

Reach/Wetland	Macrohabitat	No. sites	Month/Year of survey (monitoring period)											
			Sep. '05 (2005–06)	Jan. '06 (2005–06)	Nov. '06 (2006–07)	Dec. '07 (2007–08)	Nov. '08 (2008–09)	May '10 (2009–10)	Mar. '11 (2010–11)	Mar. '12 (2011–12)	Mar. '13 (2012–13)	Mar. '14 (2013–14)	(2014–15)	Mar. '16 (2015–16)
Murray River (adjacent to Hattah Lakes)	Riverine	3			✓	✓	✓	✓	✓	✓	✓	✓	Not sampled	✓
Chalka Creek	Anabranch	3	✓		✓			✓	✓	✓		✓		✓
Lake Mournpall	Wetland	3			✓	✓	✓	✓ *	BG	✓	✓	✓		✓
Lake Hattah	Wetland	3			✓	✓	✓	✓ *	BG	✓	✓	✓		✓
Lake Bulla	Wetland	3			✓	✓	✓		✓	✓		✓		✓
Lake Arawak	Wetland	3			✓	✓			✓	✓		✓		✓
Lake Lockie	Wetland	3		✓	✓			✓	✓	✓		✓		
Lake Yerang	Wetland	3			✓				✓	✓		✓		✓
Lake Little Hattah	Wetland	3		✓	✓			✓	✓	✓ #		✓		

*backpack electrofishing only
# netting only, at one site only, in Lake Little Hattah due to limited water.
BG = Blue-green algae bloom present

7.3.2 Statistical analyses

Species diversity index

The species diversity index is comprised of two indices: (i) α – alpha diversity (within macrohabitat diversity) and (ii) β – beta diversity (inter-macrohabitat diversity).

The α-diversity is the species richness at site level. The α-diversity was calculated as the mean (± 95% CI) number of species recorded for each site within each macrohabitat.

The β-diversity index measures species richness at macrohabitat level. It is the total number of species recorded within each macrohabitat as a proportion of the total species richness (all years) for each macrohabitat.

Extent index

The extent index is a measure of the frequency of species occurrence within macrohabitats. Extent was calculated for each sampling year as the average proportion (between 0 and 1) of sites in which each species occurred. Where extent is equal to 1, all species were present at all sites within that macrohabitat during that monitoring period.

Nativeness index

The nativeness index is the aggregate biomass of native fish species as a proportion of the total fish biomass. Most fish were measured and weighed at capture, however some fish were unable to be accurately weighed in the field. For those fish that were measured (L, mm) but not weighed, we estimated the weight ($\overline{\mathrm{W}}$, g) based on:

$$\overline{W} = 10^{[a+(b.\mathrm{Log}(\mathrm{L}/c))]}$$

Where *a* and *b* represent, respectively, the constant and slope of the exponential weight-to-length curve ($\mathrm{W} = a\mathrm{L}^{b}$) and *c* is a constant to allow conversion from fork-length or total length to standard length (Robinson 2012). Where species were abundant at a site, the first 50 individuals were measured and weighed and the remainder counted.

For each gear type and habitat type we calculated the average weight of each native and non-native species using recorded or estimated weight in each replicate (i.e. net, electrofishing shot, etc.) and counted the total number of each species in each replicate. Because unmeasured and unweighed individual fish were present in many replicates, total biomass (B') was estimated from the product of the species' mean estimated weights ($\overline{W}$) and total count (n) of all individuals. (i.e. B' = $\overline{W}$n). The biomass of native species as a proportion of the total biomass (*pNative*) was calculated for each replicate. Where a species was present in a replicate with no measured representatives in that replicate, biomass was estimated by substituting mean weight for that survey year, for that species such that; B' = species' mean weight x n.

The proportion data was transformed to normalise the error-variance using the arcsine transformation. The sampling design varies by gear type and macrohabitat (Table 7.2). For *pNative* for boat-electrofishing and small fyke nets, the lowest level of replication built into the survey design was at the 'gear' level for a nested design. Replicated sub-samples were taken from each of three sites within the single reach representing riverine or anabranch macrohabitats. The relevant model for *pNative* for the riverine and anabranch macrohabitats for the electrofishing and small fyke net gear types is:

$$pNative = \mathrm{Site} + error$$

Replicated sub-samples were taken from three sites within each water body representing wetland macrohabitat, so the relevant model for *pNative* for the wetland macrohabitat for the electrofishing and small fyke net gear types is:

pNative = Reach x Reach(Site) + *error*

For backpack electrofishing, large fyke nets, seine nets and bait traps, replicates are sub-sampled within the reach or water body (n = 8) so the 'site' classification is redundant. The relevant model for *pNative* for these gear types is:

pNative = Reach + *error*

High catch variability among sub-samples was accounted for in analysis of variance by partitioning the variance at the appropriate level when calculating the mean and variance for mean *pNative* for each macrohabitat within the icon site.

In each macrohabitat, the total variance in native fish biomass as a proportion of total fish biomass (*pNative*) was partitioned into variance among reaches and among sites within reaches, and between replicates (i.e. sub-samples from nets or electrofishing). We used the mean square error (MSE) among replicates and MSE among sites to calculate the standard error (SE) of the mean associated with each level of measurement. The following formula from Nichols *et al.* (2006) for the between-macrohabitat SE was used for the nested design, i.e. random reach selection with random sites within reaches with random sub-samples within sites:

$$SE = \sqrt{\frac{MSE(R)}{d} + \frac{MSE(S)}{de} + MSE\left(\frac{SS}{def}\right)}$$

Where R = between reaches, S = between sites and SS = between replicate sub-samples and *d*, *e* and *f* are the number of reaches, sites and replicates (nets or electrofishing shots) respectively. The sites in the study were chosen for a specific purpose and are not random; therefore, their associated standard deviation and SE are biased.

All statistical analyses were completed using R (R Development Core Team 2012), including the statistical package *lme4* (Bates et al. 2014) to perform a linear mixed effects analysis (LME) of the relationship between *pNative* and year. As the fixed effect, we entered year into the model. As random effects, we had intercepts for waterbody (e.g. wetland), as well as by-site random slopes for the effect of year. *P*-values were obtained by likelihood ratio tests of the full model with the effect of year against the model without the effect of year.

Where year was determined to be a significant effect by LME, pairwise T-tests, adjusted for multiple comparisons, were used to compare the means from each year with that of the present survey year 2015–16.

Table 7.2. Survey design structure showing reach, site and sub-sample nesting for each sampling-gear type and macrohabitat type used for monitoring the ecological condition of fish assemblages in the Hattah Lakes icon site. R = reach (or water body in the wetland macrohabitat), S = site, and ss=sub-sample (number of replicate nets or electrofishing samples).

Gear type	Macrohabitat		
	Anabranch	Riverine	Wetland
Electrofishing boat		1R/3S/12ss	
Electrofishing backpack	1R/1S/8ss		1–8 R/3S/2–3ss
Large fyke net	1R/3S/2ss		1–8R/3S/2ss
Small fyke net	1R/3S/2ss	1R/3S/2ss	1–8R/3S/2ss
Seine net	1R/3S/1ss	1R/3S/1ss	1–8R/3S/1ss
Ten bait traps	1R/1ss	1R/3S/1ss	1–8R/1ss

Fish expectedness index

This index is a measure of how many species occurred year to year compared to the expected number of species at an individual site within a macrohabitat based on the Sustainable Rivers Audit (SRA) fish expectedness method (MDBC 2007). The premise being, each icon site has a list of potential species that can occur, which has been generated from the data from the 10 years of the TLM program. For example, if the data show that a species of fish occurred in an average of seven out of nine wetlands over the 10 years of the TLM condition monitoring data set, then it may be reasonable to expect that species to occur in 78% of sites sampled in future surveys. Each species is then given a score (RC-F value) for how rare or cryptic the species is, based on a system originally determined for the SRA (MDBC 2007). This RC-F value of 1, 3 or 5 depends on the percentage of occurrence (Table 7.3). So if a species occurs in 78% of sites it would get a RC-F value of 5, meaning it is a common and abundant species. An expectedness metrics weighting (ωk) is associated with the RC-F value and reflects factors affecting detection, such as rareness, catchability (electrofishing susceptibility, net avoidance etc.), body size, etc. This is to account for the premise that not every species should be expected to be collected in every site but that most species should be collected somewhere in the macrohabitat (Robinson 2013). In the above example, the RC-F value of 5 would have an associated weighting value of 0.85 (Table 7.3). Expectedness (ωkj) is derived from the RC-F values for all species found at each macrohabitat.

Table 7.3. The RC-F fish value interpretation and associated expectedness weighting (Robinson 2012).

RC-F	Interpretation	Expectedness metrics weighting
1	Either rare or cryptic species. Expected to be collected in up to 20% of sites in the zone.	0.1
3	Locally abundant species. Expected to be collected in 20 to 70% of sites in the zone.	0.45
5	Common and abundant species. Expected to be collected in 70 to 100% of sites in the zone.	0.85
0	Native species not historically recorded in this zone. Can be included in nativeness metric calculations but not expectedness metric calculations.	0

7.4 Results

A total of 27 047 fish were sampled from Hattah Lakes and the adjacent Murray River during 2015–16 condition monitoring (Table 7.4; see section 6 of Part B of this report for further breakdown of data by site). This comprised of 14 species (10 native, 4 non-native), all of which had previously been recorded at the Hattah Lakes icon site. Of particular note is the paucity of large-bodied fish caught from the wetland and anabranch macrohabitats, with only native Bony herring and low numbers of non-native Common carp and Goldfish (*Carassius auratus*) being caught. Dwarf flathead gudgeon (*Philypnodon* macrostomus) were sampled for the second time for all years, and for the first time in the wetland macrohabitat. Numbers of the non-native Eastern mosquitofish (*Gambusia holbrooki*) in the wetland and anabranch macrohabitats were the highest recorded from all surveys. In the Murray River, the number of native Murray cod (*Maccullochella peelii*) and Golden perch were also the highest recorded for any survey period. Murray cod were numerically twice as abundant as the previous highest abundance recorded (in 2013–14). Length-frequency distribution plots for Murray cod and Golden perch caught in the Murray River are displayed in Figure 45 and Figure 46, respectively.

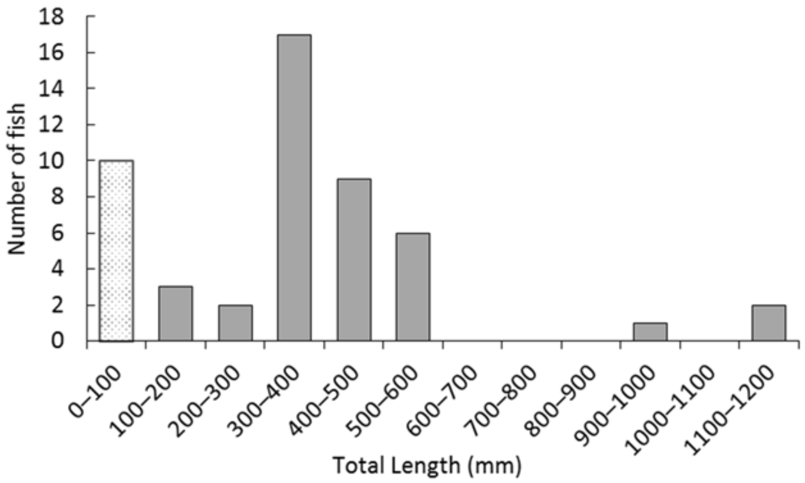


Figure 45. Length-frequency distribution of Murray cod caught in the Murray River adjacent to Hattah Lakes in 2015–16. Shaded bar represents young-of-year based on age-length studies (Anderson et al. 1992).

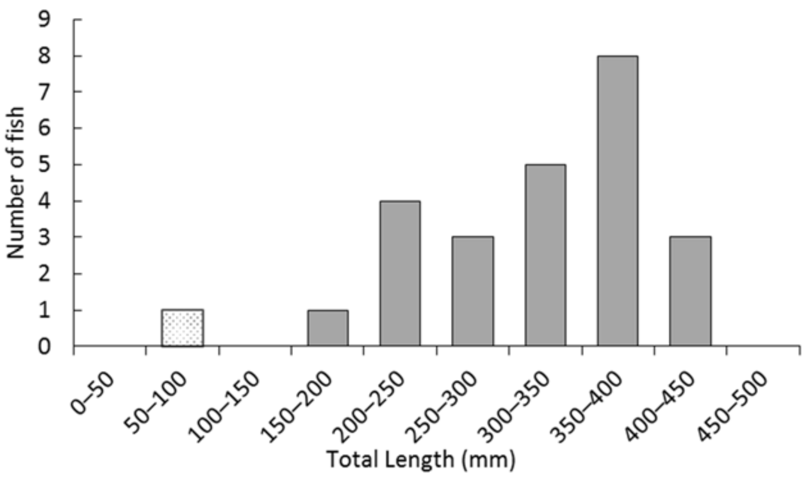


Figure 46. Length-frequency distribution of Golden perch caught in the Murray River adjacent to Hattah Lakes in 2015–16. Shaded bar represents young-of-year based on age-length studies (Mallen-Cooper & Stuart 2003).

Table 7.4. Abundance of fish species for all macrohabitats across all years.

		Native												Non-native						
		Large-bodied					Small-bodied							Large-bodied				Small-bodied		
Year	Macrohabitat	Bony herring	Golden perch	Murray cod	Silver perch	Spangled perch	Australian smelt	Carp gudgeon	Dwarf flathead gudgeon	Flathead gudgeon	Murray-Darling rainbowfish	Un-specked hardyhead	Native Total	Common carp	Common carp x Goldfish	Goldfish	Oriental weatherloach	Eastern mosquitofish	Non-native Total	Grand Total
2005–06	Anabranch	4	3	3	1		1	414		36		35	497			1			1	498
	Riverine												Not Sampled							
	Wetland						1	52					53							53
	Total	4	3	3	1		2	466		36		35	550			1			1	551
2006–07	Anabranch		4				81	66					151		1				1	152
	Riverine	11	9	6	2		114	3				1	146	20		3			23	169
	Wetland	1	61				119	6501		145			6827		1				1	6828
	Total	12	74	6	2		314	6570		145		1	7124	20	2	3			25	7149
2007–08	Anabranch												Not Sampled							
	Riverine	17	24	7	1					1			50	9					9	59
	Wetland		5				2939	12199		844			15987	2		20			22	16009
	Total	17	29	7	1		2939	12199		845			16037	11		20			31	16068
2008–09	Anabranch												Not Sampled							
	Riverine		2	2	1		1	64			2	46	118	7		1			8	126
	Wetland		1				106	1895					2002	17		111			128	2130
	Total		3	2	1		107	1959			2	46	2120	24		112			136	2256
2009–10	Anabranch	62	12		3		1	140		7	7	12	244	1					1	245
	Riverine	610	20	4	1		50	72			16	201	974	33		32		170	235	1209
	Wetland			1	7		350	1		1		1	361	5					5	366
	Total	672	32	5	11		401	213		8	23	214	1579	39		32		170	241	1820
2010–11	Anabranch	18	11				40	22171		18	4	136	22398	97		1	5	179	282	22680
	Riverine	88	10				11	11921	1	38	28	574	12671	82		4	4	4584	4674	17345
	Wetland	20	57		1	1	152	11937		135	7	553	12863	947		31	40	197	1215	14078
	Total	126	78		1	1	203	46029	1	191	39	1263	47932	1126		36	49	4960	6171	54103
2011–12	Anabranch	8	1				10	3153		4	3	36	3215	338		55	7	443	843	4058
	Riverine	126	23	1	1		23	53			5	2	234	190		3		63	256	490
	Wetland	203	17			1	4	11494		166	2	59	11946	76		93	1	777	947	12893
	Total	337	41	1	1	1	37	14700		170	10	97	15395	604		151	8	1283	2046	17441
2012–13	Anabranch												Not Sampled							
	Riverine	260	12	7	1		27	271		1	11	19	609	35		1		28	64	673
	Wetland	10						52					62	46				416	462	524
	Total	270	12	7	1		27	323		1	11	19	671	81		1		444	526	1197
2013–14	Anabranch	16	5	1			6	32			1		61	535		14		342	891	952
	Riverine	900	20	26	2		125	104			31	81	1289	143		10		16	169	1458
	Wetland	274	10	3	1		2	505		1			796	6272		32	54	1988	8346	9142
	Total	1190	35	30	3		133	641		1	32	81	2146	6950		56	54	2346	9406	11552
2014–15													Not Sampled							
2015–16	Anabranch						579	2921		2	1	1	3504	2		2	1	1889	1894	5398
	Riverine	437	31	52	2		340	43		2	96	169	1172	40		4		183	227	1399
	Wetland	57					1050	8062	12	662	1	28	9872	4		2	7	10365	10378	20250
	Total	494	31	52	2		1969	11026	12	666	98	198	14548	46		8	8	12437	12499	27047

7.4.1 Alpha species diversity

For the 2015–16 survey, alpha (native) species diversity within anabranch or riverine macrohabitats did not significantly differ from previous surveys, with the exception of 2011–12 for the anabranch macrohabitat (Figure 47). For the wetland macrohabitat the number of reaches (or wetlands) varies from year to year depending on inundation. This appears to influence α-diversity; the more reaches sampled, the greater the diversity (but also variability). The wetland macrohabitat α-diversity for 2015–16 is not significantly different to most of the other years where five of more reaches were sampled but is significantly higher than most of the years where four or fewer reaches were sampled.

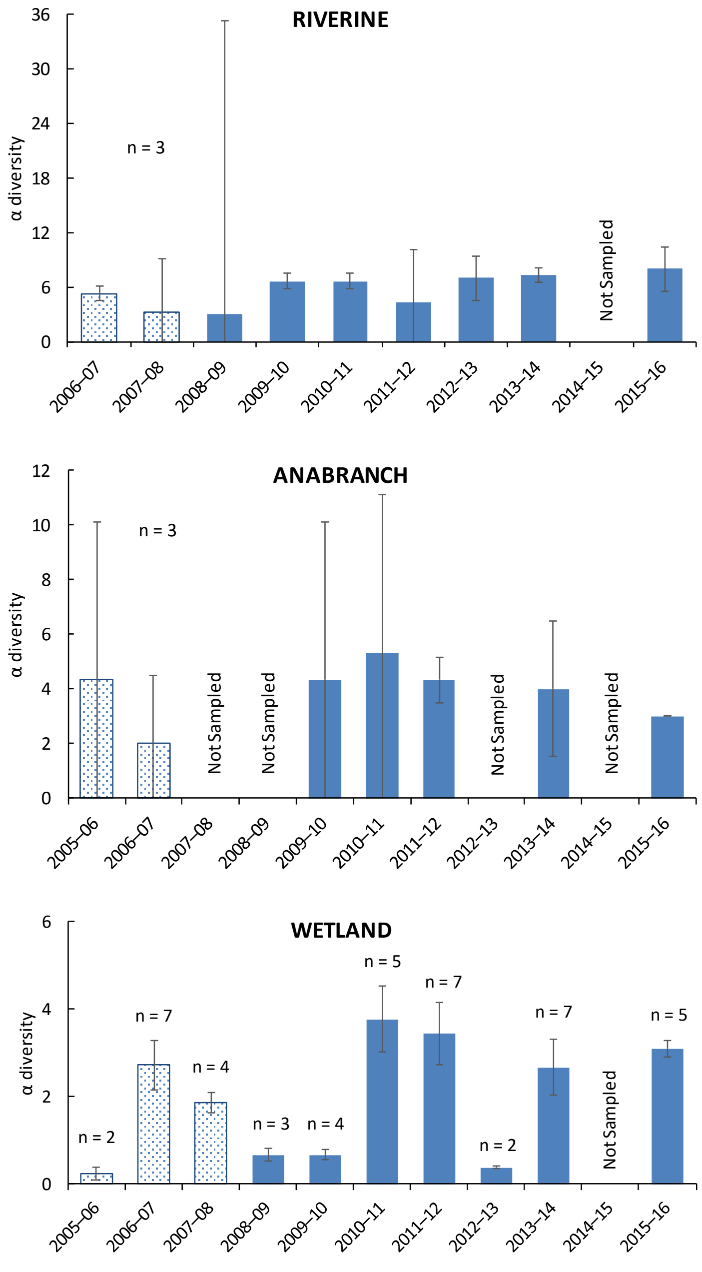


Figure 47. Mean (± 95% CI) α-diversity (i.e. within macrohabitat) of native species in the Hattah Lakes system.

7.4.2 Beta species diversity

In 2015–16 beta (native) species diversity was lowest in the anabranch macrohabitat, where just over half the expected number of species were caught (Figure 48). This number was only slightly higher in the wetland macrohabitat, where 7 out of 11 expected species were caught. In the riverine macrohabitat, the equal highest number of species were caught (n = 9) in any single monitoring period compared to total number of species across all monitoring periods (n = 10). Across all monitoring periods the β-diversity has been relatively stable in the riverine macrohabitat, while the wetland macrohabitat is the most variable.

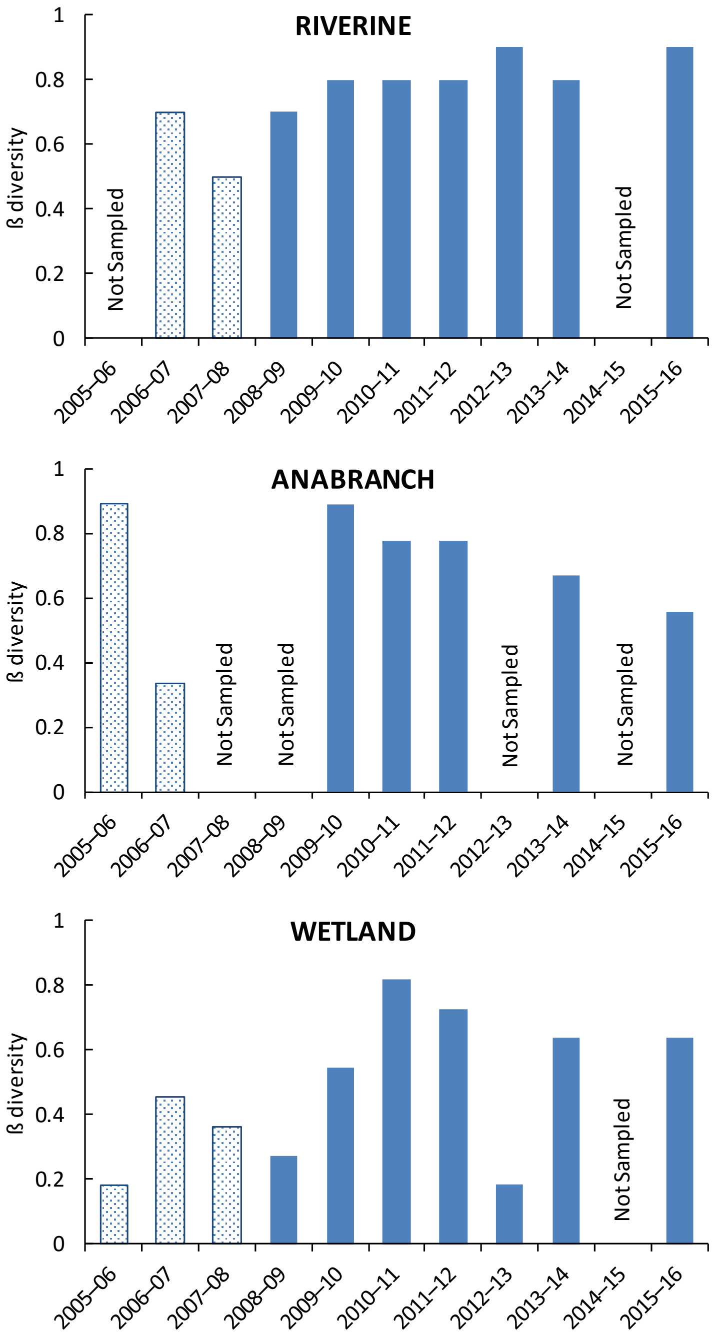


Figure 48. Beta (β) diversity (macrohabitat species richness) of species found in the Hattah Lakes system.

7.4.3 Extent

In addition to beta diversity, extent indicates the distribution of the species as a proportion of occurrence, by site, for each macrohabitat. A high amount of variability exists across all macrohabitats for all monitoring periods (Figure 49). Consequently, no significant difference (as indicated by 95% confidence intervals) can be detected for extent between years for each of the macrohabitats.

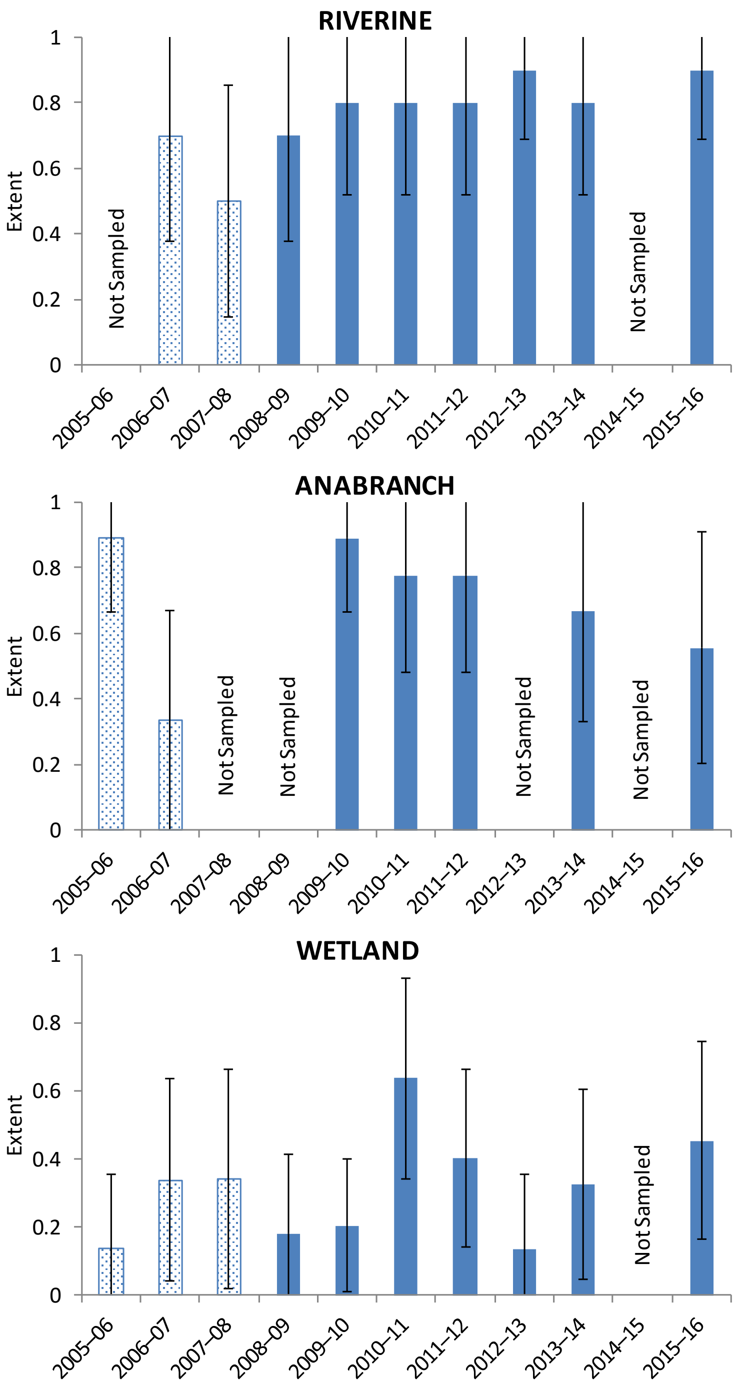


Figure 49. Mean (± 95%CI) extent of species observed in each macrohabitat, where a maximum score of 1 indicates all species were recorded at all sites within macrohabitat.

7.4.4 Nativeness

The proportions of fish biomass comprising native fish species, mean annual *pNative* (± SE) values by macrohabitat (anabranch, riverine and wetland) and gear type (small fyke nets, electrofishing, large fyke nets, seine net and bait traps); are shown in Figure 50 to Figure 54

Small fyke nets

Using small fyke nets, the proportion of native fish as biomass has varied among years surveyed for the anabranch (χ^2 (df, 5) = 37.9, $p < 0.001$), wetland (χ^2 (df, 14) = 272, $p < 0.001$) and riverine (χ^2 (df, 6) = 35.4, $p < 0.001$) macrohabitats (Figure 50.)

Pairwise T-tests show that there has been a large increase to 2015–16 in the proportion of native fish biomass since the previous survey (2013–14) in anabranch and wetland macrohabitats ($p < 0.001$). Native fish biomass remains similar to the previous survey in the riverine macrohabitat. In the wetland macrohabitat, the proportion of native fish biomass is significantly lower ($p < 0.01$–0.001) in 2015–16 than in most years prior to 2012–13 (except 2011–12) (Figure 50).

Electrofishing

Using boat electrofishing, the proportion of native fish as biomass varied among survey years for riverine macrohabitats (χ^2 (df, 8) = 24.8, $p = 0.002$), with a minimum during the high river flows of 2011–12 and maximum at the 2013–14 survey and towards the end of the millennium drought. In anabranch and wetland macrohabitats, the proportion of native fish has been highly variable among years (χ^2 (df, 4) = 47, $p < 0.001$ and χ^2 (df, 11) = 137, $p < 0.001$, respectively), with minimums between 2012 and 2014 (Figure 51).

Pairwise T-tests show that in riverine macrohabitats there has been no change to 2015–16 in the proportion of native fish since the previous survey (2013–14); while anabranch and wetland macrohabitats show strong increases ($p < 0.001$) (Figure 51).

Bait traps

In riverine and anabranch macrohabitats, the proportion of native fish biomass has remained stable at or above 0.7 for all years of record. The proportion of native fish biomass in bait trap catches in wetland macrohabitats has varied among years surveyed (χ^2 (df, 9) = 64, $p < 0.001$); and for the present survey is an increase from 2013–14 ($p < 0.001$) (Figure 52).

Large fyke nets

The proportion of native fish biomass from large fyke net catches in the wetland macrohabitat differed significantly among years (χ^2 (df, 14) = 105, $p < 0.001$), and pairwise T-tests showed a significant increase between 2013–14 and the present survey year ($p < 0.001$) (Figure 53).

In the anabranch macrohabitat the annual variability is high for the proportion of native fish biomass and differences among years are not statistically significant.

Seine nets

The proportion of native fish biomass estimated from seine net catches in the wetland macrohabitat showed an overall significant effect from year (χ^2 (df, 10) = 36.2, $p < 0.001$), and pairwise T-tests showed that there was a significant increase between the 2013–14 and present surveys ($p = 0.001$) Figure 54). For the anabranch and riverine macrohabitats, the mean proportion of native fish biomass in the present survey was high (~0.9) although high annual inter-site variability masks any significant difference (Figure 54).

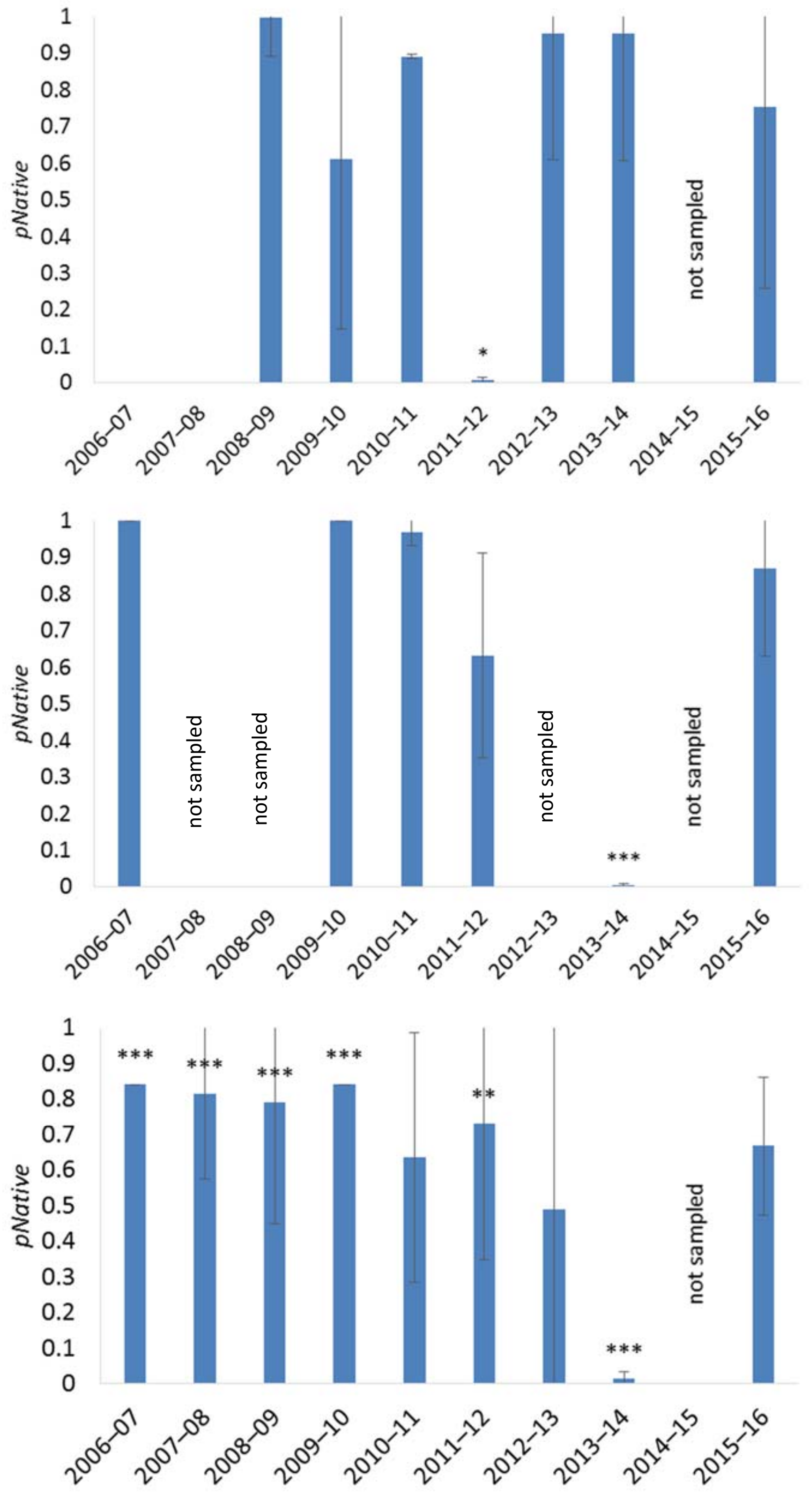


Figure 50. Mean proportion (± SE) of total fish biomass as native fish biomass (pNative) for each year sampled with small fyke nets. Top to bottom: riverine, anabranch and wetland macrohabitats. *, ** or *** denotes a significant difference (p < 0.05, p < 0.01 or p < 0.001, respectively) between the sample year and the present (2015–16) survey year.

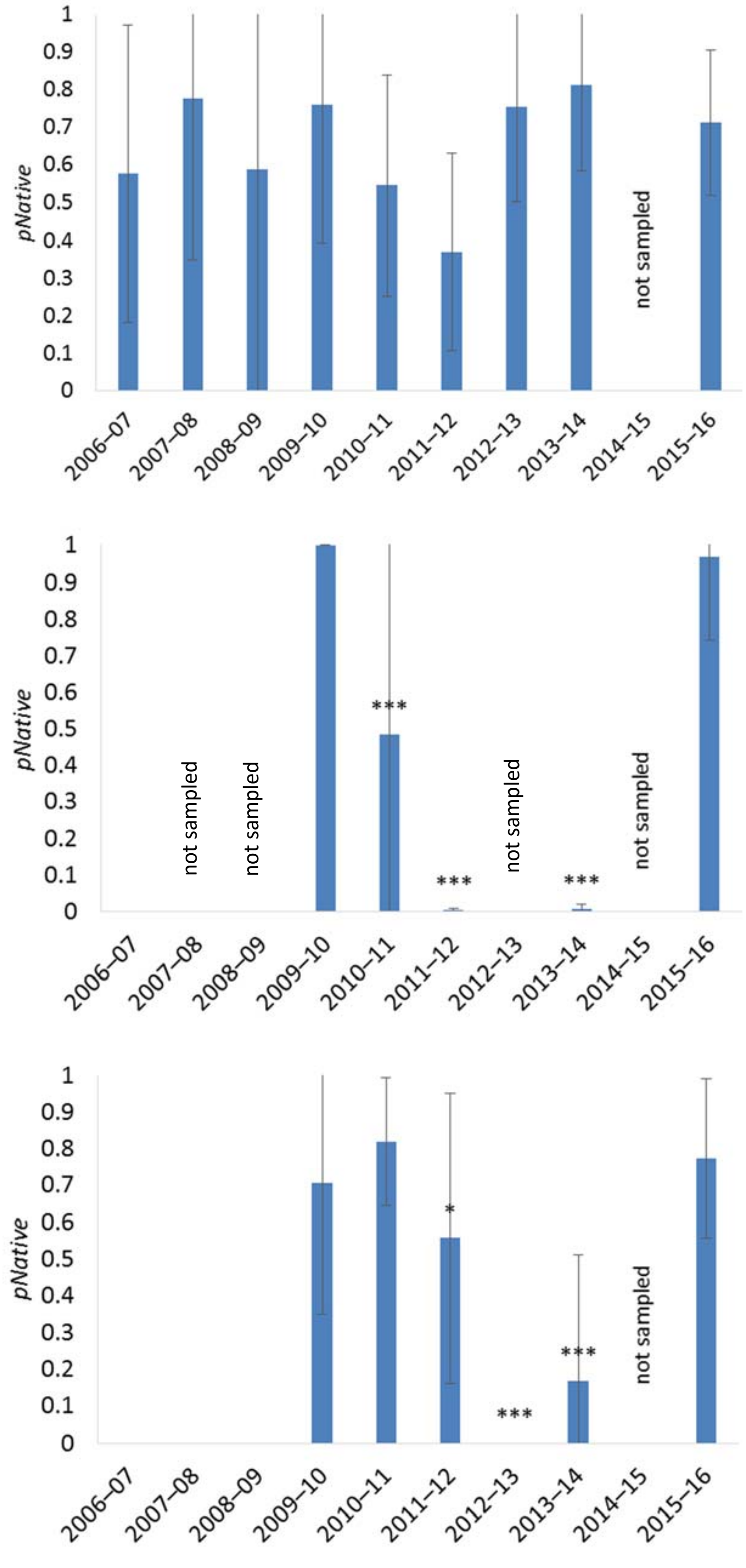


Figure 51. Mean proportion (± SE) of total fish biomass as native fish biomass (*pNative*) for each year sampled with boat electrofishing in riverine habitat (top) or back-pack electrofishing in anabranch macrohabitat (middle) or wetland macrohabitat (bottom). *or *** denotes a significant difference ($p < 0.05$ or $p < 0.001$, respectively) between the sample year and the present (2015–16) survey year.

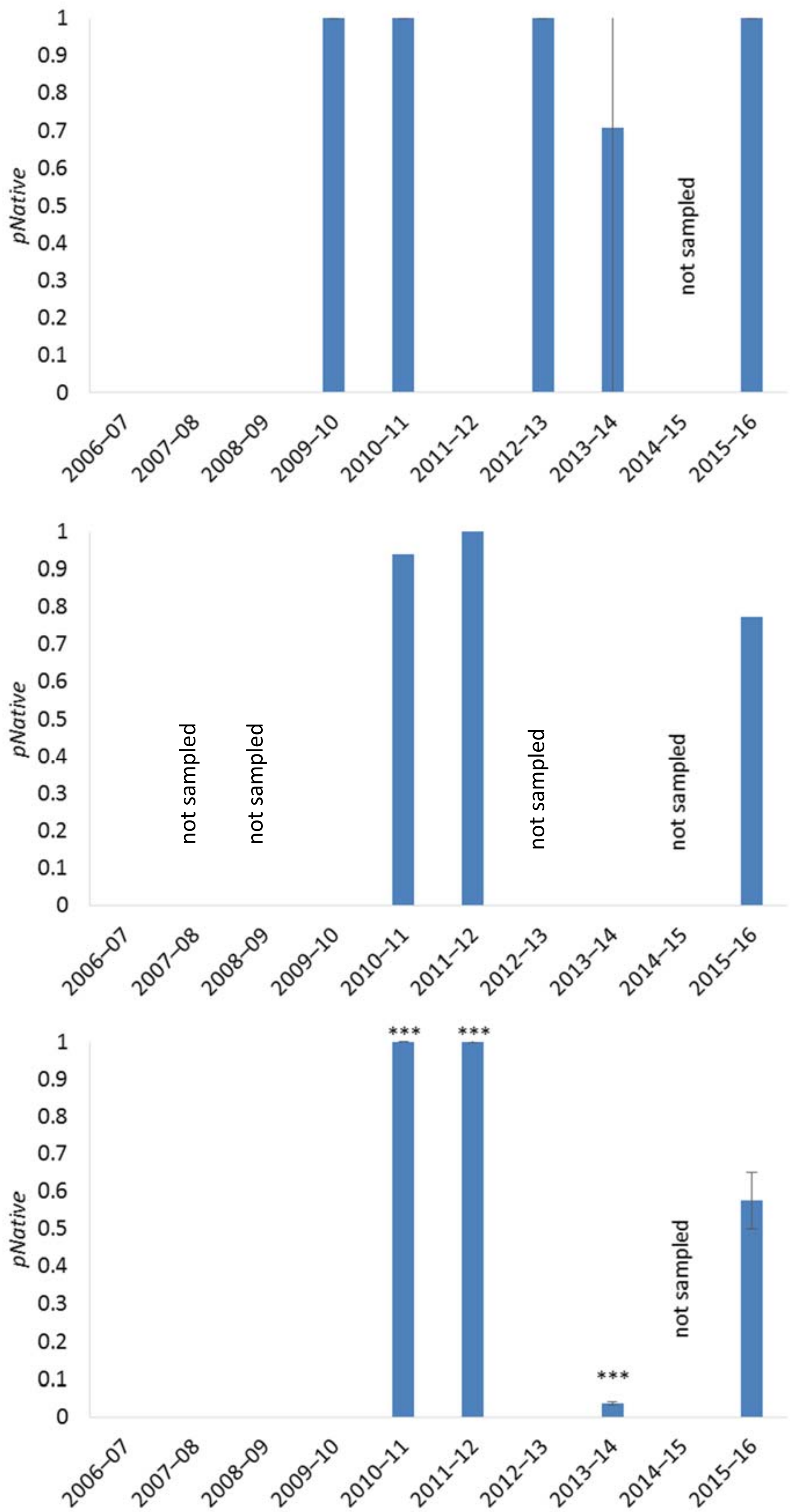


Figure 52. Mean proportion (± SE) of total fish biomass as native fish biomass (*pNative*) for each year sampled with bait traps. Top to bottom: riverine, anabranch and wetland macrohabitats. *** denotes a significant difference ($p < 0.001$) between the sample year and the present (2015–16) survey year.

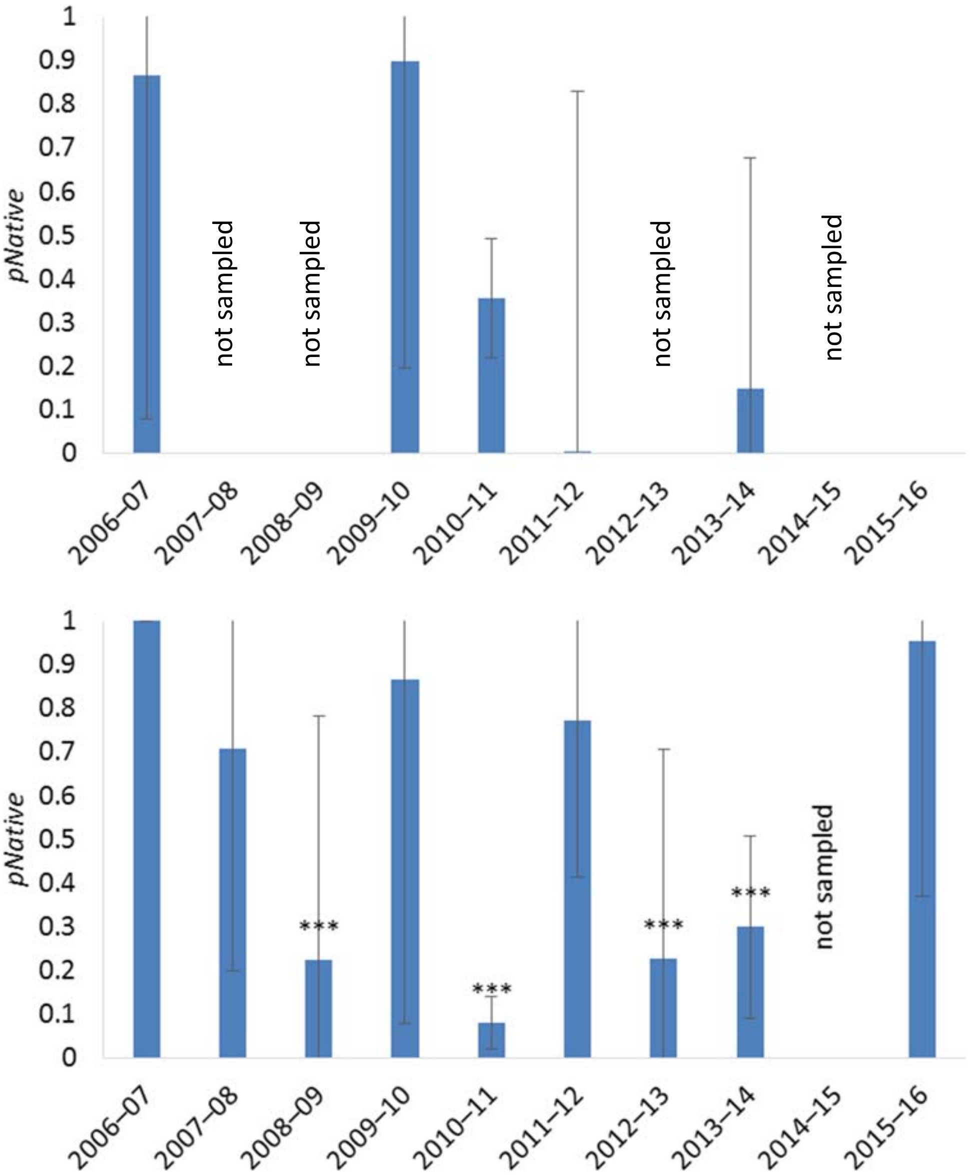


Figure 53. Mean proportion (± SE) of total fish biomass as native fish biomass (*pNative*) for each year sampled with large fyke nets in anabranch (top) and wetland (bottom) macrohabitats. *** denotes a significant difference ($p < 0.001$) between the sample year and the present (2015–16) survey year.

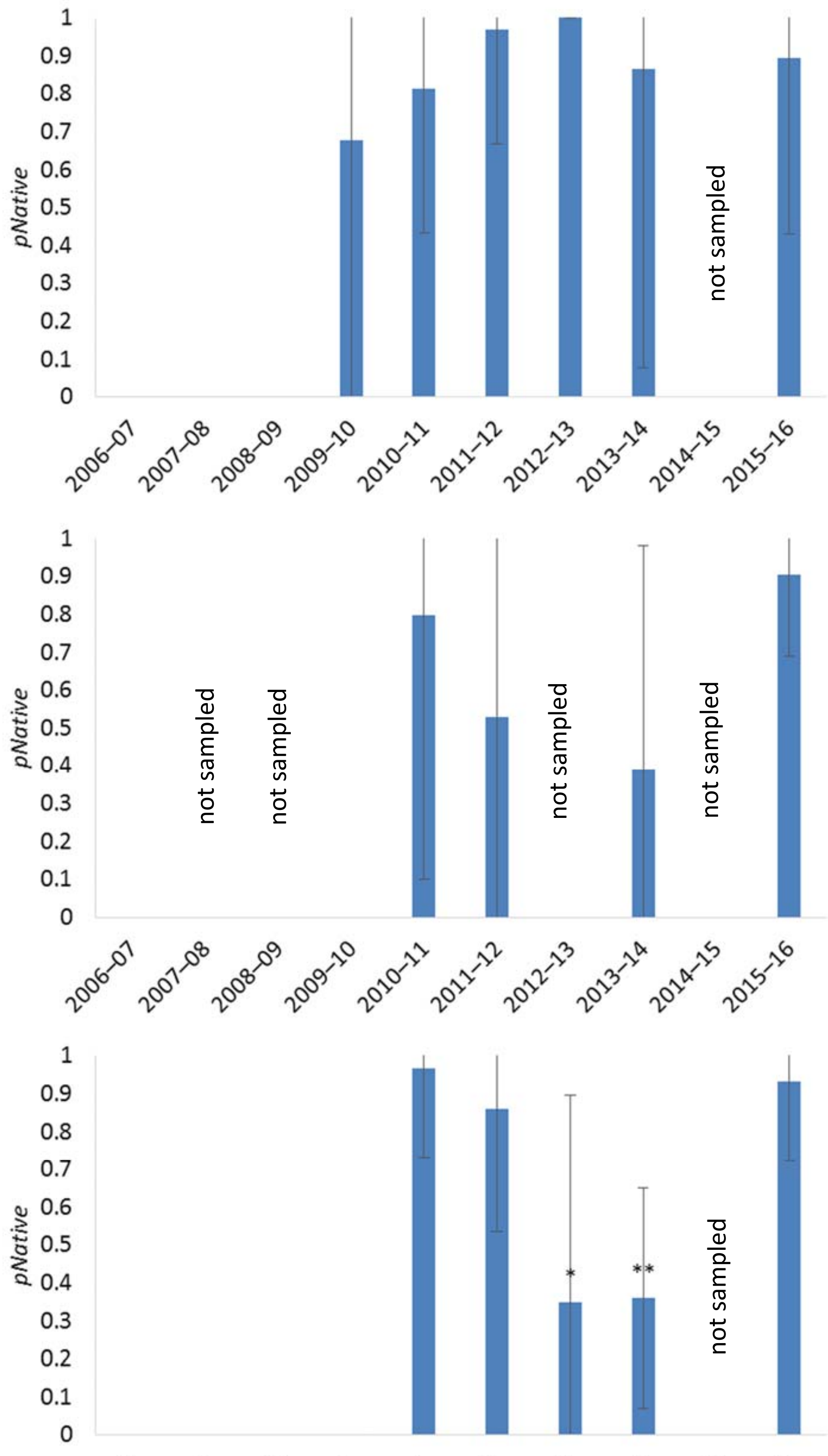


Figure 54. Mean proportion (± SE) of total fish biomass as native fish biomass (*pNative*) for each year sampled with seine nets. Top to bottom: riverine, anabranch and wetland macrohabitats. * or ** denotes a significant difference (p < 0.05 and p < 0.01, respectively) between the sample year and the present (2015–16) survey year.

7.4.5 *Fish expectedness*

In 2015–16, the average number of species caught was higher than the expected score for the icon site only in the riverine macrohabitat (Figure 55). Additionally, the average number of species recorded in the riverine macrohabitat was higher than for all other monitoring periods. Generally, the number of species recorded in each monitoring period in the riverine macrohabitat exceeds the expected score (exceptions: 2007–08, 2008–09 and 2011–12). In 2015–16, the number of species in the wetland macrohabitat, while not exceeding the icon site expected score, was the second highest recorded from all sampling years. The only monitoring period where the average number of species in the wetland macrohabitat was higher than expected was in 2010–11. Similarly, the Anabranch macrohabitat only exceeded expected values in 2010–11.

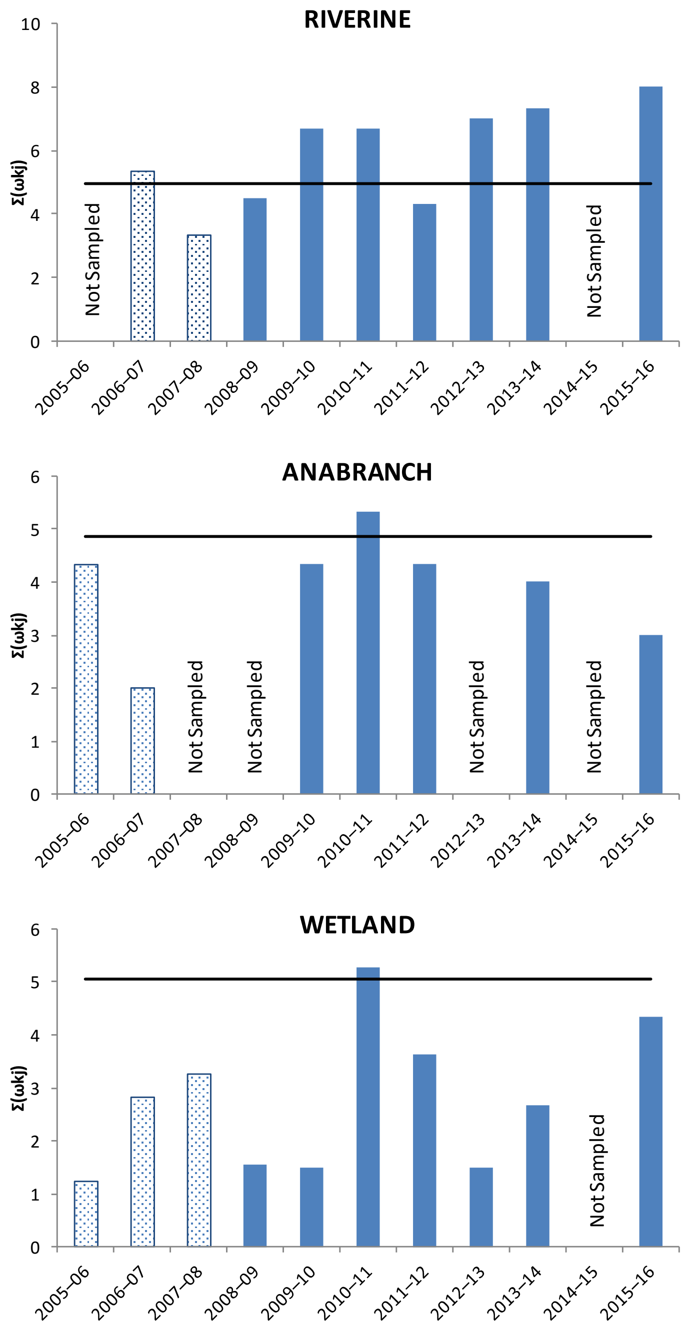


Figure 55. Average number of species observed in each macrohabitat. The horizontal line indicates the icon site RC-F expectedness Σ(ωkj) score for that macrohabitat.

7.5 Discussion

In the 2015–16 monitoring period, all but two reaches (Lakes Lockie and Little Hattah) held water and were consequently sampled. The fish community sampled was similar to other monitoring periods with no 'new' species caught. Dwarf flatheaded gudgeon, a patchily distributed species in the Murray–Darling Basin (Lintermans 2007), was caught for the second time in the region and the first time in the Hattah Lakes icon site (it had previously been caught in the Murray River adjacent to the wetlands). Spangled perch (*Leiopotherapon unicolor*) was previously detected in the Hattah Lakes, and was not recorded in 2015–16. This is unsurprising given that previous captures at Hattah Lakes followed flooding, which likely assisted dispersal of this species down the Darling River from further north (Ellis et al. 2015).

It appears that many small-bodied native wetland fish species, Carp gudgeon (*Hypseleotris* spp.), Flatheaded gudgeon (*Phylipnodon grandiceps*) and Australian smelt (*Retropinna semoni*) have thrived in the Hattah Lakes over the past few years. Numbers of these species were comparatively high compared with other surveys, suggesting successful breeding and recruitment in the Hattah Lakes. While the Hattah Lakes have received pumped water on numerous occasions over the past few years, and most recently a small flow in 2015, it is assumed that these are predominantly fish that have bred within the wetland, due to relatively low numbers of these species in the adjacent Murray River and the fact that many were probably younger than individuals pumped in during the larger flows of 2013–14.

During the 2015–16 monitoring, relatively low numbers of large-bodied non-native Common carp and Goldfish were caught in the Hattah Lakes. Two likely reasons for this are: (i) that pumping of water into the Hattah Lakes leads to a 'filtering' of the fish community (Vilizzi et al. 2013) and (ii) that as the wetlands draw down and disconnect (particularly the shallower wetlands), Common carp tend to become trapped and perish before re-filling by subsequent pumping. While very few Common carp were caught, many large individuals (> 500 mm total length) were observed by staff during work at a number of wetlands in the icon site.

The highest number of non-native Eastern mosquitofish were caught in 2015–16 compared to all other survey years in the Hattah Lakes. Their small size and tolerance for poor water quality has likely assisted their survival in shallow drying wetlands and being highly mobile assists with dispersal upon re-inundation. While it is thought that delivery of environmental water through pumps excludes this species (Vilizzi et al. 2013) they could have survived in the few pools remaining following flooding in 2010–11 and subsequent drawdowns, before re-populating the Hattah Lakes.

With the exception of Bony herring, no other large-bodied native species were sampled in the Hattah Lakes in 2015–16 condition monitoring. Fish sampling of the water entering Hattah Lakes via the pumps in 2013 and 2014 showed a number of Murray cod and Golden perch juveniles being deposited in the Hattah Lakes from the Murray River (Brown et al. 2015b; Freestone et al. 2014). A representation of these fish have since been caught during a number of different projects in the Hattah Lakes (including 2013–14 condition monitoring), with the most recent captures of both species taking place in spring 2015 (Wood & Brown 2016). While Golden perch were recorded exiting the system following this sampling (through drawdown, where water exited Hattah Lakes to the Murray River through Messengers Regulator), many remained in the system (Wood & Brown 2016). It would be safe to assume that both species are still present in the Hattah Lakes, albeit in low abundances.

Length-frequency analysis of Murray cod indicates that a large proportion of the population (20%) is comprised of young-of-year fish, indicating a strong breeding season. There is some concern about the paucity of individuals between 600 and 1000 mm total length. This range reflects the old legal-take slot for anglers. The presence of a few fish greater than 750 mm is encouraging and these fish should be now be protected as important breeding stock by the new angling legal-take slot range (550–750 mm total length).

The greatest number of Golden perch were caught in the Murray River adjacent to Hattah Lakes. However, unlike Murray cod, the population was predominantly comprised of large fish (> 150 mm total length), with only one young-of year fish caught. This is perhaps not unusual for Golden perch as riverine flooding (increases in river flow) is generally required to stimulate large-scale breeding, and this has been lacking in this region over the past few years.

7.5.1 Species diversity

The Hattah Lakes have experienced relatively high variability in species composition and distribution across all sites and all monitoring periods. The species composition and distribution in the Murray River by comparison has been much more stable across the same time periods. This reflects the general stability of the water levels in the river as compared to the extremes experienced in the Hattah Lakes.

Environmental factors play a significant role in species composition in ephemeral systems, such as the Hattah Lakes. The main driving factor in ephemeral systems is the constantly changing water level, which has many flow-on effects such as changing water quality, habitat type, food resources and predation pressure. Consequently, it is difficult to make sound comparisons between years, as it is rare for the wetlands to ever be in a similar ecological state from survey year to survey year.

As water draws down in the wetlands and Chalka Creek, habitat contracts and water quality deteriorates. This normally results in the death of native species in the first instance, while non-native species, which can survive poorer water quality conditions (Kennard et al. 2005; Koehn 2004; Wilson 2006), last for longer in the residual pools. Consequently, native species are generally in low abundance or absent in wetlands that have drawn down considerably.

At the other end of the watering spectrum, flooding, particularly natural, results in higher species diversity and greater distribution across a wetland complex. This would be expected, as the sink (wetlands) and source (river) are a homogenous system allowing free fish passage. Natural flooding can also boost overall species diversity in the region by increasing longitudinal distribution of vagrant species. This occurred for two species during natural flooding in the Murray–Darling system in 2010–11. Non-native Oriental weatherloach (*Misgurnus anguillicaudatus*), previously recorded at Robinvale (~150 river kilometres upstream) as its easterly most distribution, were caught at Hattah Lakes during the flood (and have been since). Spangled perch were also caught at Hattah Lakes the same year, having dispersed from further north (their southern limit is generally regarded as Menindee Lakes) down the Darling River.

The second method of filling, pumped water delivery (flooding of Hattah Lakes only), greatly influences the fish community. Pumping of water to the Hattah Lakes filters the fish community in a number of ways. Pumping is size-selective; only small fish can be entrained through the pump and survive. Also, certain species may be excluded by their habitat preference (e.g. the pump off-take is located deep in the water column, whereas Eastern mosquitofish generally inhabit the surface of the water column). This means that not all species present in the source water (i.e. Murray River) are transferred to the Hattah Lakes.

It is considered that natural connection, which allows unrestricted passage between the Murray River and Hattah Lakes, is the best watering method for achieving high species diversity and even distribution of species throughout the Hattah Lakes system.

7.5.2 Nativeness

The metric *pNative* may be a reasonably sensitive indicator of ecological condition in relation to the specified objectives for fish at the Hattah Lakes icon site. In approximately half the cases (i.e. gear x macrohabitat combinations) there were significant differences among years surveyed that indicated variation in the proportion of native fish as biomass. Also, for half of the cases considered, the 2015–16 survey marks a statistically significant increase in this index in comparison with the 2013–14 survey.

Native fish biomass as a proportion of total fish biomass should be broadly correlated with breeding success and increased relative abundance of native fish (i.e. recruitment) when that success is at the expense of non-native fish breeding success. A low index could be expected following strong recruitment of non-native fish species and poor native fish recruitment, whereas a high index should represent strong recruitment of native fish species and poor recruitment of non-natives.

Native small-bodied fish species should produce annual recruitment in Hattah Lakes water bodies; however, two of the most common non-native fish recorded at Hattah Lakes, Common carp and Eastern mosquitofish, show strong population responses to inundation of floodplain habitat. Fishing gear that is selective for small-bodied species and juvenile large-bodied species is likely to show the most immediate and strongest response.

Proportion of native fish as biomass estimated for wetlands using small fyke nets was relatively stable and high (> 0.8) during the period when water levels were maintained by environmental water delivery via pumps until 2010–11. This period was characterised by the pumped delivery of a 'filtered' fish community comprised mainly of native species, with very few non-native species when Common carp abundance, particularly juveniles and larvae, was generally relatively low in the Murray River following the prolonged drought. The widespread natural flooding in 2010 and 2011 was followed by declining *pNative* values in the wetland macrohabitat, driven mainly by strong year-classes of non-native Common carp, until wetlands dried in 2012. The delivery of pumped environmental water, which filled the majority of wetlands in summer 2013–14, was accompanied by large numbers of small Common carp (Brown et al. 2015b), leading to a minimum value in the *pNative* metric for the 2013–14 survey. Two years on, the present survey shows that the relative proportion of native fish biomass has again increased across most macrohabitats, despite record numbers of the small-bodied alien species Eastern Mosquitofish. Common carp abundance has crashed, suggesting weak recruitment following the environmental flows of 2013–14 and 2014–15. The recovery in native fish biomass in the wetland and anabranch macrohabitats is due to the numerical dominance of native Carp gudgeon, Australian smelt and Un-specked hardyhead (*Craterocephalus fulvus*), and was boosted in the riverine samples due to relatively high abundance of Murray cod.

Large fyke nets are more selective for large-bodied fish. The *pNative* metric using large fyke net data from the wetland macrohabitat declines from almost 100% native biomass in 2006–07 to a minimum in 2010–11. The strong increase following natural flooding in 2010–11 was lagged by a year, suggesting significant recruitment and growth of large-bodied native fish following the wetland inundation a year earlier. Common carp grow more rapidly than most native large-bodied species (Vilizzi & Walker 1999). The decline in *pNative* in 2012–13 is consistent with a fish community where the biomass of Common carp became dominant due to growth over 1–2 years following successful breeding and subsequent recruitment to the gear. Mid-sized large-bodied fish are generally susceptible to this sampling gear and the return to high nativeness values in the present survey of the wetland macrohabitat is a good indicator of relatively low Common carp abundance again.

Linear mixed modelling is appropriate for ongoing analyses of the whole dataset where year, reach, and site are a mixture of fixed and random effects and time-series effects with repeated measures, and where an unbalanced design results from missing values occurring due to sites having been omitted, for example during drier times (Crawley 2013). For future monitoring, increased power to detect change would result from increasing the number of reaches sampled in the macrohabitat categories riverine and anabranch, and ensuring adequate replication at all sites for all gears.

7.5.3 Progress toward the objectives

The overarching objective for fish in the Hattah Lakes is:

Maintain high-quality habitat for native fish in wetlands and support successful breeding events.

The first more detailed objective is:

Increase distribution, number and recruitment of local wetland fish—including hardyhead, Australian smelt and gudgeon—by providing appropriately managed habitat.

For the Hattah Lakes wetland macrohabitat, numbers of local wetland fish were relatively high in 2015–16 compared with previous years. Their ongoing presence in the Hattah Lakes indicates recruitment. Distribution of species across wetlands containing water was relatively homogenous during 2015–16. However, in an ephemeral system, such as Hattah Lakes, it cannot be expected that the distribution and number of local wetland fish can increase indefinitely. With another year of no inflows, a significant decline in local wetland fish numbers and distribution would be expected as wetlands dry.

The second more detailed objective is:

Maximise use of floodplain habitat for recruitment of all indigenous freshwater fish.

The only floodplain habitat currently at the Hattah Lakes icon site exists in wetlands (i.e. lakes). The wetlands are currently in a drying phase, which is resulting in an overall decline in habitat availability. The ongoing presence of small-bodied fish between sampling years indicates that recruitment has been occurring for these species.

Silver perch, Golden perch and Murray cod species do not breed within the Hattah Lakes wetlands. Their eggs and juveniles are transported to the wetland and anabranch macrohabitats by flows from the Murray (pumped or natural). The wetland and anabranch macrohabitats within the icon site service as 'nursery habitat' for these largely riverine species. So the spatial context for the term 'recruitment' for long-lived native fish species like Golden perch, Murray cod and Silver perch is really a basin-scale one (King et al. 2009). To maximise the use of floodplain habitats for recruitment (as stated in the objective) means to make use of the nursery habitat within the Hattah Lakes icon site for the greater population in the lower Murray River. Presently the sampling strategy for TLM condition monitoring does not address this objective for all species, but only for small-bodied species (wetland specialists) that may recruit via breeding *within* the icon site. Recent TLM intervention monitoring has been developing the knowledge-base and evaluating operation methods for watering infrastructure to achieve this objective for large-bodied fish species such as Golden perch (Brown et al. 2014; Brown et al. 2015b; Wood & Brown 2016). The detailed objective *Maximise use of floodplain habitat for recruitment of all indigenous freshwater fish* is only achievable if a methodology can be developed for managing the environmental watering infrastructure that enables large-bodied flow-dependent species (i.e. Golden perch and Silver perch) that have developed as juveniles within the lakes to emigrate to the Murray River as they mature.

8 References

Anderson AJ, Gorley RN, Clarke KR (2008) *PERMANOVA+ for PRIMER: Guide to Software and Statistical Methods.* PRIMER-E Ltd, Plymouth, UK.

Anderson JR, Morison AK, Ray DJ (1992) Age and growth of Murray cod, *Maccullochella peelii* (Perciformes: Percichthyidae), in the lower Murray–Darling Basin, Australia, from thin-sectioned otoliths. *Australian Journal of Marine and Freshwater Research* **43**, 983–1013.

Anderson M (2005) PERMANOVA: a FORTRAN computer program for permutational multivariate analysis of variance. Department of Statistics, University of Auckland, Auckland, New Zealand.

Bates D, Maechler M, Bolker B, Walker S (2014) lme4:Linear-mixed effects models using Eigen and s4. R package version 1.1-6.

Baumgartner LJ, Conallin J, Wooden I, Campbell B, Gee R, Robinson WA, Mallen-Cooper M (2014) Using flow guilds of freshwater fish in an adaptive management framework to simplify environmental flow delivery for semi-arid riverine systems. *Fish and Fisheries*, **15**, 410-427.

BOM (2016) Climate data online. In. Bureau of Meteorology. Available from: http://www.bom.gov.au/climate/data/. Accessed: 27/5/2016

Briggs S, Maher M (1983) Litter fall and leaf decomposition in a river red gum (*Eucalyptus camaldulensis*) swamp. *Australian Journal of Botany* **31**, 307–316.

Brock M, Casanova M (1997) Plant life at the edge of wetlands: ecological responses to wetting and drying patterns. In: *Frontiers in Ecology: Buliding the links* (eds. Klomp N, Lunt I), pp. 181–192. Elsevier Science Ltd, Oxford, UK.

Brock MA, Capon SJ, Porter JL (2006) Disturbance of plant communities dependent on desert rivers. In: *Ecology of Desert Rivers* (ed. Kingsford R). Cambridge University Press, Sydney, NSW.

Brown P, Ellis I, Henderson M, Lampard B (2014) Chapter 2: Fish community response. In: *Response of fish and vegetation communities to the operation of The Living Murray environmental regulators at Hattah Lakes* (eds. Freestone F, Brown P, Henderson M,Campbell C, Ellis I, Lampard B and Cranston G), p. 8–26. Murray–Darling Freshwater Research Centre, Mildura, Vic.

Brown P, Henderson M, Freestone F, Campbell C (2015a) The Living Murray condition monitoring refinement project report updates for the Lindsay–Mulcra–Wallpolla Islands and Hattah Lakes icon sites 2014–15. MDFRC Publication 84/2015. Report to the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre, Mildura, Vic.

Brown P, Huntley S, Ellis I, Henderson M, Lampard B (2015b) Movement of fish eggs and larvae through the Hattah Lakes environmental pumps. MDFRC Publication 50/2015. Report to the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre and La Trobe University, Mildura, Vic.

Brown P, Sivakumaran KP, Stoessel D, Giles A (2005) Population biology of carp (*Cyprinus carpio* L.) in the mid-Murray River and Barmah Forest Wetlands, Australia. *Marine and Freshwater Research* **56**, 1151–1164.

Butcher R, Hale J (2011) Ecological character description for Hattah–Kulkyne Lakes Ramsar site. Report to the Department of Sustainability, Environment, Water, Population and Communities. Canberra, ACT.

Campbell CJ, Johns CV, Nielsen DL (2014) The value of plant functional groups in demonstrating and communicating vegetation responses to environmental flows. *Freshwater Biology* **59**, 858–869.

Capon SJ (2005) Flood variability and spatial variation in plant community composition and structure on a large arid floodplain. *Journal of Arid Environments* **60**, 283–302.

Catford JA, Jansson R (2014) Drowned, buried and carried away: effects of plant traits on the distribution of native and alien species in riparian ecosystems. *New Phytologist* **204**, 19–36.

Chong C, Walker KF (2005) Does lignum rely on soil seed bank? Germination and reproduction phenolongy of *Muehlenbeckia florulenta* (Polygonaceae). *Australian Journal of Botany* **53**, 407–415.

Craig AE, Walker KF, Boulton AJ (1991) Effects of edaphilic factors and flood frequency on the abundance of Lignum (*Muehlenbeckia florulenta* Meissner) (Polygonaceae) on the River Murray Floodplain, South Australia. *Australian Journal of Botany* **39**, 431–443.

Crawley MJ (2013) *The R Book*, 2nd edition. Wiley, Chichester, United Kingdom.

Cunningham GM, Mulham WE, Milthorpe PL, Leigh JH (1992) *Plants of western New South Wales.* Inkata Press, Marrickville, NSW.

Ecological Associates (2007) Feasibility investigation of options for the Hattah Lakes. Report prepared for the Mallee Catchment Management Authority by Ecological Associates.

Ellis I, Whiterod N, Linklater D, Bogenhuber D, Brown P, Gilligan D (2015) Spangled perch (*Leiopotherapon unicolor*) in the southern Murray–Darling Basin: Flood dispersal and short-term persistence outside its core range. *Austral Ecology* **40**, 591–600.

Ellis I, Wood D (2011) Assessment of the fish community in Lake Kramen (North West Victoria) upon re-inundation after a 17 year dry spell. Report to the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre, Mildura, Vic.

EPA, MDFRC (2008) Implications of pumping and ponding water on water quality and the development of diverse aquatic ecosystems – Intervention monitoring of the Hattah Lakes Icon Site 2006/07. Report to the Murray–Darling Basin Commission by the Environmental Protection Agency Victoria and The Murray–Darling Freshwater Research Centre, Mildura, Vic.

Freestone F, Brown P, Henderson M, Campbell C, Ellis I, Lampard B, Cranston G (2014) Response of fish and vegetation communities to the operation of The Living Murray environmental regulators at Hattah lakes. MDFRC Publication 26/2014. Report to the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre, Mildura, Vic.

George A, Walker K, Lewis M (2005) Population status of eucalypt trees on the River Murray floodplain, South Australia. *River Research and Applications* **21**, 271–282.

George AK (2004) *Eucalypt regeneration on the Lower Murray floodplain, South Australia.* PhD Thesis, The University of Adelaide, Adelaide, SA.

Harden G (1992) *Flora of New South Wales Volume 3.* UNSW Press, Sydney, NSW.

Harden G (1993) *Flora of New South Wales Volume 4.* UNSW Press, Sydney, NSW.

Harden G (2000) *Flora of New South Wales Volume 1.* UNSW Press, Sydney, NSW.

Harden G (2002) *Flora of New South Wales Volume 2*, Revised edn. UNSW Press, Sydney, NSW.

Henderson M, Freestone F, Cranston G, Campbell C, Vlamis T, Huntly S, Brown P (2014) The Living Murray Condition Monitoring at Hattah Lakes 2013–14: Part A – Main Report. Report to the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre, Mildura, Vic.

Henderson M, Vilizzi L, Cranston G, Wood D (2012) The Living Murray Condition Monitoring at Hattah Lakes 2011/12: Part B – Supplementary Material. Report to the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre, Mildura, Vic.

Holland KL, Tyerman SD, Mensforth L, Walker G (2006) Tree water sources over shallow, saline groundwater in the lower River Murray, south-eastern Australia: implications for groundwater recharge mechanisms. *Australian Journal of Botany* **54**, 193–205.

James CS, Capon SJ, Quinn GP (2015) Nurse plant effects of a dominant shrub (*Duma florulenta*) on understorey vegetation in a large, semi-arid wetland in relation to flood frequency and drying. *Journal of Vegetation Science* **26**, 985–994.

James CS, Capon SJ, White MG, Rayburg SC, Thoms MC (2007) Spatial variability of the soil seed bank in a heterogeneous ephemeral wetland system in semi-arid Australia. *Plant Ecology* **190**, 205–217.

Jensen A (2008) *The role of seed banks and soil moisture in recruitment of semi-arid floodplain plants: The River Murray, Australia.* PhD Thesis, The University of Adelaide, Adelaide, SA.

Jolly ID, Walker GR (1996) Is the field water use of *Eucalyptus largiflorens* F. Muell. affected by short-term flooding? *Australian Journal of Ecology* **21**, 173–183.

Jolly ID, Walker GR, Thorburn PJ (1993) Salt accumulation in semi-arid floodplain soils with implications for forest health. *Journal of Hydrology* **150**, 589–614.

Junk W, Bayley P, Sparks R (1989) The flood pulse concept in river-floodplain systems. Proceedings of the International Large River Symposium. *Canadian Special Publication of Fisheries and Aquatic Sciences* **106***,* 110–127.

Junk W, Wamtzem K (2004) The flood pulse concept: new aspects, approaches, and applications—an update In *Second International Symposium on the Management of Large Rivers for Fisheries* (pp. 117-149). Food and Agriculture Organization and Mekong River Commission, FAO Regional Office for Asia and the Pacific.

Kennard MJ, Arthington AH, Pusey BJ, Harch BD (2005) Are alien fish a reliable indicator of river health? *Freshwater Biology* **50**, 174–193.

King A, Ramsey D, Baumgartner L, Humphries P, Jones P, Koehn J, Lyon J, Mallen-Cooper M, Meredith S, Vilizzi L, Ye Q, Zampatti B (2009) Environmental requirements for managing successful fish recruitment in the Murray River valley – Review of existing knowledge. *Arthur Rylah Institute for Environmental Research Technical Report Series No. 197*, Department of Sustainability and Environment. Melbourne, Australia.

Koehn JD (2004) Carp (*Cyprinus carpio*) as a powerful invader in Australian waterways. *Freshwater Biology* **49**, 882–894.

Leblanc M, Tweed S, Van Dijk A, Timbal B (2012) A review of historic and future hydrological changes in the Murray–Darling Basin. *Global and Planetary Change* **80–81**, 226–246.

Lintermans M (2007) *Fishes of the Murray–Darling Basin: An introductory guide.* Murray–Darling Basin Commission, Canberra, ACT.

Lyon J, Stuart I, Ramsey D, O'Mahony J (2010) The effect of water level on lateral movements of fish between river and off-channel habitats and implications for management. *Marine and Freshwater Research* **61**, 271–278.

Mac Nally R, Parkinson A, Horrocks G, Conole L, Tzaros C (2001) Relationships between terrestrial vertebrate diversity, abundance and availability of coarse woody debris on south-eastern Australian floodplains. *Biological Conservation* **99**, 191–205.

Maher MT, Braithwaite LW (1992) Patterns of waterbird use in the wetlands of the Paroo, a river system of inland Australia. *The Rangeland Journal* **14**, 128–142.

Maheshwari B, Walker K, McMahon T (1993) The impact of flow regulation on the hydrology of the River Murray and its ecological implications. University of Melbourne and University of Adelaide, Mighty Mouse Publishing Services, Melbourne, Vic.

Maheshwari BL, Walker KF, McMahon TA (1995) Effects of regulation on the flow regime of the River Murray, Australia. *Regulated Rivers: Research and Management* **10**, 15–38.

Mallen-Cooper M, Stuart IG (2003) Age, growth and non-flood recruitment of two potamodromous fishes in a large semi-arid/temperate river system. *Regulated Rivers: Research and Management* **19**, 697–719.

McCarthy B, Tucker M, Vilizzi L, Campbell C, Walters S (2009) Implications of pumping water on the ecology of Hattah Lakes. Report to the Murray–Darling Basin Commission by The Murray–Darling Freshwater Research Centre, Mildura, Vic.

MDBA (2010) Fish condition monitoring approach (v.2). Murray–Darling Basin Authority, Canberra, ACT.

MDBA (2012a) Assessment of environmental water requirements for the proposed Basin Plan: Hattah Lakes. Murray–Darling Basin Authority, Canberra, ACT.

MDBA (2012b) Ground-based survey methods for The Living Murray assessment of condition of river red gum and black box populations. Murray–Darling Basin Authority, Canberra, ACT.

MDBA (2012c) Hattah Lakes: Environmental water management plan 2012. Murray–Darling Basin Authority, Canberra, ACT.

MDBC (2005) The Living Murray foundation report on the significant ecological assets targeted in the first step decision. Murray–Darling Basin Commission, Canberra, ACT.

MDBC (2006) The Hattah Lakes icon site environmental management plan 2006–2007. Murray–Darling Basin Commission, Canberra, ACT.

MDBC (2007) Sustainable Rivers Audit protocols – Approved manual for implementation period 4: 2007–08. Murray–Darling Basin Commission, Canberra, ACT.

Council MDBM (2003) Murray–Darling Basin Ministerial Council communiqué, 14 November 2003. *Murray–Darling Basin Commission, Canberra*.

MDFRC (2011) The Living Murray: Condition Monitoring Program design for Hattah Lakes, p. 121. A report prepared for the Murray-Darling Basin Commission by The Murray-Darling Freshwater Research Centre.

Nichols SJ, Robinson WA, Norris RH (2006) Sample variability influences on the precision of predictive bioassessment. *Hydrobiologia* **572**, 215-233. Nicol J, Weedon J (2006) Understorey vegetation monitoring of the Chowilla River Red Gum watering trials. Aquatic Sciences, South Australian Research and Development Institute, Adelaide, SA.

Overton I, McEwan K, Gabrovsek C, Sherrah J (2006) The River Murray floodplain inundation model (RiM-FIM): Hume Dam to Lower Lakes. CSIRO, Canberra, ACT.

Pettit NE, Froend RH (2001) Variability in flood disturbance and the impact on riparian tree recruitment in two contrasting river systems. *Wetlands Ecology and Management* **9**, 13–25.

R Development Core Team (2012) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Reid M, Quinn G (2004) Hydrologic regime and macrophyte assemblages in temporary floodplain wetlands: Implications for detecting responses to environmental water allocations. *Wetlands* **24**, 586–599.

Roberts J, Marston F (2011) *Water regime for wetland and floodplain plants: a source book for the Murray–Darling Basin.* National Water Commission, Canberra, ACT.

Robinson W (2012) Calculating statistics, metrics, sub-indicators and the SRA Fish theme index. A Sustainable Rivers Audit Technical Report. Murray-Darling Basin Authority, Canberra.

Robinson W (2013) The Living Murray: Towards assessing whole of icon site condition. In: *Report to the Murray Darling Basin Authority*, p. 122.

Robinson W (2014a) The Living Murray condition monitoring plan refinement project: summary report. Technical report to the Murray–Darling Basin Authority, December 2014. 75 pp.

Robinson W (2014b) The Living Murray condition monitoring plan refinement project: technical document for sensitivity and power analyses of whole of icon site condition assessment. Report to the Murray–Darling Basin Authority, 294 pp.

Rogers K, Ralph T (2011) *Floodplain wetland biota in the Murray–Darling Basin: water and habitat requirements.* CSIRO Publishing, Collingwood, Vic.

Sainty GR, Jacobs SW (1981) Waterplants of New South Wales. *New South Wales: Water Resources Commission 550p.-illus., col. illus., maps, keys.. En Icones, Maps, Chromosome numbers. Geog* **7**.

Scholz O, Reid J, Wallace T, Meredith S (2007a) The Living Murray Initiative: Lindsay–Mulcra–Wallpolla Islands and Hattah Lakes icon sites condition monitoring program design. MDFRC Publication 1/2007. Report to the Mallee Catchment Management Authority by The Murray-Darling Freshwater Research Centre, Mildura, Vic.

Scholz O, Sharpe C, Fraser P, Henderson M, Ellis I (2007b) The Living Murray Initiative: Lindsay–Mulcra–Wallpolla Islands and Hattah Lakes icon sites 2006–7 condition monitoring program data. MDFRC Publication 2/2007. Report to the Mallee Catchment Management Authority. by The Murray–Darling Freshwater Research Centre, Mildura, Vic.

SKM (2003) Hattah Lakes integrated water management plan. Report to the Mallee Catchment Management Authority by Sinclair Knight Mertz Pty Ltd, Armadale, Vic.

Snowball D (2001) Eucalyptus largiflorens *F. Muell. (Black Box) as an indicator of an extreme palaeoflood event at the River Murray at Overland Corner, South Australia.* Honours Thesis, University of South Australia, Adelaide, SA.

Souter N (2005) Flood regime change in the Hattah Lakes Victoria resulting from regulation of the River Murray. *Transactions of the American Fisheries Society* **129**, 74–80.

Stein A, Gerstner K, Kreft H (2014) Environmental heterogeneity as a universal driver of species richness across taxa, biomes and spatial scales. *Ecology Letters* **17**, 866–880.

Stuart I, Jones M (2006) Large, regulated forest floodplain is an ideal recruitment zone for non-native common carp (*Cyprinus carpio* L.). *Marine and Freshwater Research* **57**, 333–347.

Thoms M, Beyer P, Rogers K (2006) Variability, complexity and diversity: the geomorphology of river ecosystems in dryland regions. In: *Ecology of Desert Rivers* (ed. Kingsford R), pp. 47-75.

Vilizzi L, McCarthy B, Scholz O, Sharpe C, Wood D (2013) Managed and natural inundation: benefits for conservation of native fish in a semi-arid wetland system. *Aquatic Conservation: Marine and Freshwater Ecosystems* **23**, 37–50.

Vilizzi L, Walker KF (1999) Age and growth of the common carp, *Cyprinus carpio*, in the River Murray, Australia: validation, consistency of age interpretation, and growth models. *Environmental Biology of Fishes* **54**, 77–106.

Walsh N, Entwisle T (1994) *Flora of Victoria Volume 2.* Inkata Press, Melbourne Vic.

Walsh N, Entwisle T (1996) *Flora of Victoria Volume 3.* Inkata Press, Melbourne Vic.

Warwick NWM, Brock MA (2003) Plant reproduction in temporary wetlands: the effects of seasonal timing, depth, and duration of flooding. *Aquatic Botany* **77**, 153–167.

Wilson G G (2006) Impact of invasive exotic fishes on wetland ecosystems in the Murray–Darling Basin. In *Native fish and wetlands of the Murray–Darling Basin: action plan, knowledge gaps and supporting papers Proceedings of a workshop held in Canberra ACT* (pp. 7-8).

Wood D, Brown P (2016) Lateral fish movement at the Hattah Lakes icon site. MDFRC Publication 95/2016. Report to the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre, Mildura, Vic.

Young WJ (2001) *Rivers as Ecological Systems: The Murray–Darling Basin.* Murray–Darling Basin Commission, Canberra, ACT.