The Living Murray: Annual condition monitoring at Lindsay-Mulcra-Wallpolla Icon Site 2017–18: Part A

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The Living Murray: Annual Condition Monitoring at Lindsay-Mulcra-Wallpolla Icon Site 2017–18. Part A

Final report prepared for the Mallee Catchment Management Authority by the School of Life Sciences Albury–Wodonga and Mildura.

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Executive summary

This report details the condition monitoring undertaken at Lindsay-Mulcra-Wallpolla (LMW) in 2017–18 as part of The Living Murray (TLM) Condition Monitoring Program 2006–18. 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 TLM scheme. Specifically it assesses whether sustainable native fish, birds and vegetation communities are being maintained across icon sites.

This report documents inter-annual changes in whole-of-icon-site condition by presenting structured chapters for each of the ecological components monitored: River Red Gum, Black Box, wetland vegetation, floodplain vegetation, Lignum, fish and birds.

This survey year (2017–18) experienced relatively low flows and did not result in over-bank flooding. During this period, environmental water was delivered to some locations across the LMW icon site.

Below is a ‘Report card’ (Table 1.1) which highlights each of the ecological objectives, for each component and indicates if the objective has been achieved, partially achieved or not achieved. The foundation for assigning the objective outcome is based (where applicable) on the result of targets and indices applied to each component.

Table 1.1 Report card indicating if the ecological objective for each component has been achieved, partially achieved or not achieved.

<table>
<thead>
<tr>
<th>Component</th>
<th>Objective</th>
<th>Achieved</th>
<th>Partially achieved</th>
<th>Not achieved</th>
</tr>
</thead>
<tbody>
<tr>
<td>River Red Gum</td>
<td>Sustainable populations of River Red Gum</td>
<td></td>
<td></td>
<td>X</td>
</tr>
<tr>
<td>Black Box</td>
<td>Sustainable populations of Black Box</td>
<td></td>
<td></td>
<td>X</td>
</tr>
<tr>
<td>Wetland vegetation</td>
<td>To maintain and restore a mosaic of healthy floodplain communities</td>
<td></td>
<td>X</td>
<td></td>
</tr>
<tr>
<td>Floodplain vegetation</td>
<td>To maintain and restore a mosaic of healthy floodplain communities</td>
<td></td>
<td></td>
<td>X</td>
</tr>
<tr>
<td>Lignum</td>
<td>Improve condition of Lignum communities</td>
<td></td>
<td></td>
<td>X</td>
</tr>
<tr>
<td>Cumbungi</td>
<td>Limit Cumbungi growth</td>
<td></td>
<td></td>
<td>X</td>
</tr>
<tr>
<td>Fish</td>
<td>Maintain native fish populations, their relative abundance and diversity</td>
<td></td>
<td></td>
<td>X</td>
</tr>
<tr>
<td>Birds</td>
<td>Provide occasional breeding and roosting habitat for colonial waterbirds</td>
<td></td>
<td></td>
<td>X</td>
</tr>
<tr>
<td></td>
<td>Provide habitat suitable for migratory birds, especially species listed</td>
<td></td>
<td></td>
<td>X</td>
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<tr>
<td></td>
<td>under the JAMBA, CAMBA and RoKAMBA agreements (between Australia and Japan,</td>
<td></td>
<td></td>
<td>X</td>
</tr>
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<td></td>
<td>China and the Republic of Korea respectively).</td>
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<td></td>
<td>X</td>
</tr>
</tbody>
</table>
River Red Gum

The highest frequency of River Red Gums with a crown extent score of ≥ 4 was recorded at LMW in 2017–18 indicating improved condition. The target of 85% of River Red Gums attaining a crown extent score ≥ 4 was achieved during 2017–18.

The River Red Gum population was dominated by seedlings during the 2015–18 monitoring round, although they were not as numerous as in 2012–15. Between the last two monitoring periods there is evidence of growth as there has been a shift upward in tree size classes. The population status index score is yet to attain the threshold of 79% for any monitoring period.

As only one of the targets for River Red Gum was achieved, the ecological objective; sustainable populations of River Red Gums at LMW icon site is only partially being met.

Black Box

This year (2017–18) represents the highest percentage of trees achieving a crown extent score of ≥ 4 across all monitoring periods. Consequently, the target for crown condition was achieved for 2017–18.

The population status of Black Box at LMW is of some concern. While some seedlings were present in 2017–18, their numbers were comparatively low and the population was dominated by larger trees. The low proportion of juvenile trees and misshaped population curve reflects a low population status index that does not achieve the minimum threshold for 2015–18.

Overall, the ecological objective relating to sustainable populations of Black Box at the LMW icon site is only partially being met.

Wetland vegetation

Most wetlands at LMW had dried after being inundated in November 2016. However, three wetlands retained water, supporting a spatial and temporal mosaic of habitats at LMW. In response to drying conditions, there was an increase in the proportion of drought tolerant plant species and a decrease in the proportion of terrestrial damp and amphibious vegetation. Therefore, the overarching vision for wetlands at LMW to maintain a mosaic of healthy wetland communities is being met in 2017–18.

Floodplain vegetation

Water responsive plant species at sites of the LMW floodplain that are often or sometimes-flooded show some continued benefit after being flooded in November 2016. Overall, however, water responsive species richness and abundance has decreased since 2016–17. The rarely-flooded areas remain predominantly long-dry and dominated by drought tolerant species. Therefore, the overarching vision to maintain healthy floodplain communities at LMW is partially being met.

The Whole-of-icon site score was calculated for the entire time series at LMW floodplains for the first time this year (2017–18). No change in water responsive species richness and abundance was detected over the whole icon site since TLM surveys began. However, it should be noted that the score reflects plant responses to flooding events.
**Lignum**

Monitoring of Lignum during 2017–18 indicates there has been an improvement in condition for Lignum Swamp and Woodland strata. However, only Lignum Woodland achieved the target of ≥70% plants with a Lignum Condition Index score ≥4. Conversely, Lignum Shrubland did not show an improvement in condition during this monitoring period. At the icon site scale, Lignum condition improved but failed to reach the target value. As such, the icon site Condition Index increased from 0.375 in 2016–17 to 0.667 in 2017–18.

**Cumbungi**

Surveys from 2017 indicate that Cumbungi distribution has been dramatically reduced at all reaches since it was last assessed in 2015. During 2017–18, Cumbungi was recorded at four reaches and distribution was highest in Wallpolla Creek (0.74%). As no point of reference currently exists for Cumbungi, distribution is reported against the adopted ecological objective of *limit Cumbungi growth*. Due to the reduction of Cumbungi from the previous survey, this objective is currently being met at LMW.

**Fish**

During condition monitoring sampling for 2017–18 the second highest number of fish was sampled from the LMW icon site. This included 15 species (10 native, 5 non-native), all which had previously been sampled from LMW. The community was numerically dominated by Carp gudgeon (*Hypseleotris spp.*), appearing in their highest number across all surveys. Both Golden perch (*Macquaria ambigua*) and Murray cod (*Maccullochella peelii*) were sampled in their second greatest numbers. The majority of Murray cod were recruits, something that has not occurred at LMW across any condition monitoring survey previously. Conversely, all but one individual Golden perch were adults.

Species richness, as a function of mean proportion of sites that exceed the reference level of expected species-richness, was down from the previous monitoring year (2016–17). Both the proportion of native fish recruits and the mean proportion of fish biomass had increased from the previous year. However, neither exceeded the reference values for 2017–18. As such, the objective *maintain native fish populations, their relative abundance and diversity* was only partially met for 2017–18 at LMW icon site.

**Birds**

Bi-annual waterbird monitoring at LMW during 2017–18 saw waterbird abundance higher in autumn 2018 than in spring 2017. Many wetlands were dry or close to dry, thus harbouring low or no waterbirds. Waterbird counts were highest at Lake Wallawalla on both occasions, with Australian Wood Duck dominating the community on both sampling occasions. Freckled Duck, listed as Endangered in Victoria, was also recorded during the autumn 2018 survey at Lake Wallawalla. Across the rest of LMW wetlands, Grey Teal were the most commonly recorded species.

This year’s assessments did not provide evidence of colonial breeding but did show roosting around wetlands by such species. Therefore, the objective, *provide occasional breeding and roosting habitat for colonial waterbirds* has partially been met.

No transcontinental migratory birds were observed during 2017–18; therefore’ the objective *provide habitat suitable for migratory birds, especially species listed under the JAMBA, CAMBA and RoKAMBA agreements (between Australia and Japan, China and the Republic of Korea respectively)* has not been met.
1 Introduction

1.1 Purpose of the report

This report details the condition monitoring undertaken at Lindsay, Mulcra and Wallpolla Islands (LMW) as part of The Living Murray (TLM) Condition Monitoring Program 2006–18. This work was conducted by La Trobe University, School of Life Sciences for the Mallee Catchment Management Authority (CMA). This report is a deliverable requirement for Contract No. 001631 between La Trobe University and the Mallee CMA.

1.2 Report structure

The LMW Condition Monitoring report for 2017–18 consists of two parts: Part A comprises the main report and Part B the supplementary material. Part A contains the structured reports for the ecological components monitored (River Red Gum, Black Box, wetland vegetation, floodplain vegetation, Lignum, fish and birds). 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 LMW have largely remained consistent for the duration of the program. Data analysis and reporting has benefited from a process of continuous-improvement that commenced in July 2013 (Robinson 2013), was ongoing during 2013–16 (Brown et al. 2015; Robinson 2014a, b) and drove a series of refinements. Data collection and sampling design for most ecological components has remained consistent. The exception being the Lignum condition monitoring component, which in 2016–17 adopted new methods. No condition monitoring was funded in the 2014–15 survey period; the only sampling that took place during this ‘gap’ was for tree structure and this has been included where possible.

1.3 Lindsay–Mulcra–Wallpolla Islands

The Chowilla Floodplain and LMW Islands icon site is one of six sites identified as ecologically significant as part of The Living Murray’s first step (Figure 1.1)(MDBMC 2003). The icon site comprises 43 856 ha of floodplain spanning three states: South Australia, Victoria and New South Wales (Figure 1.2). The Victorian component, LMW Islands, covers 26 156 ha in north-west Victoria, downstream of the Murray–Darling junction at Wentworth. The LMW floodplain is comprised of a variety of landforms including wetlands, billabongs, flood runners and permanently inundated anabranches. Lindsay Island, Wallpolla Island and Lake Wallawalla are listed under the Directory of Important Wetlands (Environment Australia 2001) and are nationally significant (ANCA 1996). A detailed information base for LMW Islands is provided in Chapter 5 of The Living Murray Foundation Report (MDBC 2005).

1.3.1 Hydrology

Drought conditions prevailed at LMW Islands leading up to the commencement of The Living Murray Condition Monitoring Program in 2006–07 (Figure 1.3). These conditions continued through the first five years of monitoring, until 2010–11 when above-average rainfall in the Murray–Darling Basin caused high flows in the Murray and Darling Rivers. Flows at Lock 9 peaked at 74 500 ML day$^{-1}$ in February 2011 inundating large areas of the LMW floodplain. Flows subsided in the ensuing months before steadily rising again to 49 200 ML day$^{-1}$ in April 2012. A recession and further peak occurred in August 2012 (48 000 ML day$^{-1}$). Flows returned to base levels early in 2013.
In June 2016, the Murray River at Lock 9 started rising again, reaching a peak of 82 570 ML day\(^{-1}\) in December and falling to base levels in March 2017 (Figure 1.3). During the 2016–17 survey year, LMW experienced the largest flow event in the history of The Living Murray Program. Unlike the flood in 2010–11, this recent event did not follow a decade of drought (MDBA 2017).
This report presents observations recorded the year following significant natural inundation of large areas of the icon site. This survey year (2017–18) experienced relatively low flows that did not result in any over-bank flooding. During this period, environmental water was delivered to some locations across the icon site; Lake Wallawalla (8 000 ML, September-November 2017), Sandy Creek (400 ML, November 2017), Wallpolla Horseshoe (400 ML, November 2017), Wallpolla East (600 ML, November 2017) and Webster’s Lagoon (late 2017).
2 River Red Gum

AUTHOR: DAVID WOOD

2.1 Introduction

River Red Gum (*Eucalyptus camaldulensis*) is widespread throughout the Murray–Darling Basin. Common along water courses and frequently inundated areas of the floodplain, Red Gums play an important functional role through carbon cycling and the provision of habitat (Baldwin 1999; Briggs & Maher 1983; Briggs et al. 1997; MDBC 2003). River Red Gum is a structurally dominant species within floodplain vegetation communities at Lindsay-Mulcra-Wallpolla (LMW) Islands.

Flooding is an integral part of River Red Gum ecology and provides an important source of water to sustain populations. Changes in flooding regime and groundwater status threaten the condition, recruitment and long term sustainability of Red Gum populations, 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 Red Gum communities at LMW through environmental works and the delivery of environmental water.

River Red Gum are monitored on an annual basis as outlined in the Condition Monitoring Program Plan for LMW icon site (Huntley et al. 2016a). Condition monitoring reports on River Red Gum condition and population status at the icon site scale, with reference to specific targets and indices.

2.2 Ecological objectives

Ecological objectives for LMW Islands have been in refinement since interim objectives were first developed by the 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, modelling and consultation activities over ten years (MDBA 2012b). The ecological objective for River Red Gum is:

*Maintain current condition and extent of River Red Gum communities to sustain species assemblages and processes typical of such woodland.*

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

*Sustainable populations of River Red Gums*

2.3 Method

Two methods were employed to assess the condition of River Red Gums at LMW: tree condition monitoring and population status.

Comprehensive detail on tree condition monitoring and size-class distribution methods are available in the Condition Monitoring Program design for the LMW icon site (Huntley et al. 2016a).

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 2012a). 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 2012a).
Twenty-seven sites (nine at each island), each comprising of 30 River Red Gum trees were established in 2007–08 and sampled annually to 2017–18 (with the exception of 2010–11 as the program did not run due to flooding and 2014–15).

To compensate for the loss of sample trees due to mortality, for each live tree lost, a replacement was selected (the closest live tree). Accordingly, only the live tree component of the sample set for any given year are considered when comparing inter-year differences in tree condition. For more detailed information on site establishment, locations and sampling refer to (Huntley et al. 2016a).

### 2.3.2 Population status

Population size-class distribution surveys are used to inform population status assessments. These assessments are used to evaluate long-term sustainability of River Red Gum at LMW and relate closely to the objective of maintaining extent (MDBA 2012b).

Size-class distribution of River Red Gum is assessed on a three-year rolling cycle (or “monitoring round”) such that for each year, approximately one third of sites are sampled. Transects were established in 2006–09 covering 57.3 ha, which represents approximately 1.7% of the extent of River Red Gum at LMW Islands. For more detailed information on site establishment, locations and sampling refer to Huntley et al. (2016a).

Each transect was navigated end-to-end using a hand-held GPS. The DBH of each River Red Gum tree within the transects was 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.

Sites first surveyed during 2006–09 and reassessed in 2009–12, 2012–15 and 2015–18 monitoring rounds 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

Suitable indices and associated points of reference for reporting on the condition and maintenance of River Red Gum have recently been developed (Brown et al. 2015; Robinson 2014a). This report incorporates these updated measures to evaluate and report on River Red Gum condition at LMW.

#### 2.4.1 Tree Condition

The target developed for River Red Gum condition at LMW is:

- 85% of trees with crown extent score ≥ 4

Crown extent is a visual assessment of the percentage of tree crown containing live leaves. A crown extent score of equal to or greater than four is associated with a tree crown that is greater than 40% foliated. This point of reference is based on The Living Murray (TLM) condition monitoring data collected 2007–13 that indicates River Red Gum trees with less than 40% foliated crown are at significantly higher risk of mortality (MDFRC, unpublished data).

The percentage of sampled trees with a crown extent score ≥ 4 was calculated per site and averaged across all sites. The mean is the estimate of the percentage of trees within the population with a crown extent score ≥ 4. The standard error of the mean is expressed in plots as error bars (± SE).
2.4.2 Population status

The population status index is based on the ‘inverse J-shaped’ curve (George et al. 2005) that 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 across the icon site. The metric of comparison used was Spearman’s Rho ($\rho$), which was then converted to an index value of between zero and one:

- $\rho = \frac{\sum (x_i - \bar{x})(y_i - \bar{y})}{\sqrt{\sum (x_i - \bar{x})^2 \sum (y_i - \bar{y})^2}}$
- $\text{Index} = (\rho + 1)/2$

Where $x$ and $y$ are the ranks of the observed data, and inverse J-curve DBH size-classes.

For population status, a minimum threshold reference point of a mean population status index value of 0.79 has been adopted for River Red Gum at LMW (Brown et al. 2015). This is based on previous data and is in line with values achieved toward the end of the millennium drought.

2.5 Results

2.5.1 Tree condition

The highest frequency of River Red Gums with a crown extent score of ≥ 4 was recorded at LMW in 2017–18 (Figure 2.1). This follows the trend of continual improvement since 2015–16, which is indicative of improving condition. The target of 85% of Red Gums attaining a crown extent score ≥ 4 was achieved during 2017–18, as it has every year of monitoring since 2011–12.
The Living Murray: Annual condition monitoring at Lindsay-Mulcra-Wallpolia Icon Site 2017–18. Part A

2.5.2 Population status

Smaller sized trees (0–15 cm DBH) dominated the River Red Gum population at LMW during the 2015–18 monitoring round (Figure 2.2). While not as numerous as in 2012–15, this size-class makes up more than 88% of the population in 2015–18. In 2012–15 this smallest size-class was mainly comprised of newly germinated seedlings (<2 cm DBH); however, it appears that by 2015–18 many of these seedlings had grown (Figure 2.3). This is evident as trees in larger size classes (2–5 cm DBH) increased in the number of. Additionally, during 2015–18 there was further germination of seedlings at LMW.

The population status index score is yet to attain the threshold of 79% for any monitoring round across sampling (Figure 2.4). There has been little change in score between the two most recent rounds, with large confidence intervals suggesting that a high degree of variability exists across the population.
2.6 Discussion

Following significant flooding in late 2016, there has been no subsequent overbank flooding or application of environmental water across LMW during 2017. As the previous monitoring (2016–17) was undertaken relatively soon after flooding in late 2016, the trees may not have fully responded to the inundation. By this monitoring period, the trees would have previously reached their full potential, without further inundation, thus being recorded in good condition.

The shift-up in size classes of relatively newly germinated trees is a positive trend for River Red Gum populations. With high numbers of seedlings germinating during 2012–15 and the subsequent decline due to mortality, it is positive to see that a number of these trees have established themselves and are growing. Following flooding in late 2016, it was expected that high numbers of seedlings may germinate. This did not appear to be the case with only moderate numbers recorded during this monitoring period (compared with earlier germinations). It may be that higher germination levels are evident in subsequent years following improved condition (and higher flower and seedling production) caused by inundation, such as followed flooding in 2010–11.
Large variability in the River Red Gum population exist at LMW. This is likely a result of myriad of factors including inundation history, past land uses (e.g. logging), grazing pressures (e.g. native versus stock) and other environmental factors (e.g. soil types, groundwater). This is not something that can be ‘improved’ with short-term fixes. Changes in population dynamics for long-lived species such as River Red Gum occur over many decades and even centuries (Platt et al. 1988).

For 2017–18, the target relating to more than 85% of River Red Gums with a Crown extent score of ≥ 4 has been achieved, while the minimum threshold for the population status index has not been met. Therefore, the ecological objective, sustainable populations of River Red Gums at LMW icon site is only partially being met.
3 Black box

AUTHOR: DAVID WOOD

3.1 Introduction

Black box (Eucalyptus largiflorens) is a common species at Lindsay–Mulcra–Wallpolla (LMW) and one of only a few large tree species. Black box generally occur higher on the floodplain (i.e. less frequently inundated) 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 is adapted 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 maintain the condition and extent of Black Box communities at LMW through environmental works and the delivery of environmental water. Although management options to influence Black Box at LMW Islands are currently limited, additional benefit may be achieved through the indirect influence of small scale flooding on groundwater or through the ‘piggybacking’ of environmental water on high flows.

Black box are monitored on an annual basis as outlined in the Condition Monitoring Program Plan for Chowilla-LMW icon site (Huntley et al. 2016a). Condition monitoring reports on Black Box condition and population status at the icon site scale, with reference to specific targets and indices.

3.2 Ecological objective

Ecological objectives for LMW have been in refinement since interim objectives were first developed by the 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, modelling and consultation activities over ten years (MDBC 2006). The ecological objective for Black Box is:

*Improve condition to sustain species assemblages and processes typical of Black Box woodland*

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

*Sustainable populations of Black Box*

3.3 Method

In order to address objectives relating to Black Box, two methods were employed to assess the condition of Black Box at LMW: tree condition monitoring and population status.

Comprehensive detail on tree condition monitoring and size-class distribution assessments are provided in the Condition Monitoring Program design for the LMW icon site (Huntley et al. 2016a).
3.3.2  **Tree Condition**

Twenty-seven sites each comprising of 30 Black Box trees were established in 2007–08 and sampled annually to 2017–18, except for 2010–11 due to flooding and 2014–15 as the program did not run. See Section 2.3.1 for more details on tree condition sampling method.

3.3.3  **Population status**

Size-class distribution of Black Box is assessed on a three-year rolling cycle (or “monitoring round”) such that for each year approximately one third of sites are sampled. Transects were established in 2006–09 covering 96.6 ha, which represents approximately 1.64% of the extent of Black Box at LMW. See Section 2.3.2 for more details on population status sampling method.

3.4  **Indices and points of reference**

Suitable indices and associated points of reference for reporting on the condition and maintenance of Black Box have recently been developed (Brown et al. 2015; Robinson 2014a). This report incorporates these updated measures to evaluate and report on Black Box condition at LMW.

3.4.1  **Tree Condition**

The target developed for Black Box condition at LMW icon site is:

- 80% of trees with crown extent score ≥ 4

See Section 2.4.1 for more detail on development of points of reference for tree condition.

3.4.2  **Population status**

A minimum threshold reference point of a mean population status index value of 0.80 has been adopted for Black Box population status at LMW (Brown et al. 2015). This is based on previous data and is in line with values achieved toward the end of the millennium drought. See Section 2.4.2 for more detail on development of points of reference for population status.

3.5  **Results**

3.5.1  **Tree condition**

Following an increase in the frequency of trees with a crown extent score ≥ 4 between 2015–16 and 2016–17, this trend continued to this monitoring period (2017–18) (Figure 3.1). This represents the highest percentage of trees achieving this score across all monitoring periods. Consequently, the target of more than 80% of trees meeting a crown condition score of ≥ 4 was achieved for 2017–18, as has been the case since 2011–12.

3.5.2  **Population status**

The population status of Black Box at LMW is of some concern. Currently the population is dominated by trees of the 15–30 cm DBH size-class and to a lesser extent, 30–45 cm DBH class (Figure 3.2). Ideally, in a model sustainable population, the smallest size class should be the most dominant, with declining numbers in subsequent size classes that approximates an inverse ‘J-shaped’ curve. While some seedlings are present in 2017–18, their numbers are comparatively low (Figure 3.3).
Figure 3.1 Mean frequency (± SE) of trees with crown extent scores ≥ 4 recorded for Black Box at sites sampled annually in summer between 2008–18 (except for 2010–11 and 2014–15)(target value = 80%).

The low proportion of juvenile trees and misshaped population curve reflects a low population status index that does not achieve the minimum threshold for 2015–18 (Figure 3.4). The population status index for Black Box at LMW has not been achieved in any monitoring round. Large confidence intervals also indicate high variability across the Black Box population at LMW.

Figure 3.2 Size-class distribution of live Black Box trees (0–225 cm DBH) at LMW; n = 3925 in 2006–09, n = 3951 in 2009–12, n = 4082 in 2012–15, n = 3842 in 2015–18.

Figure 3.3 Size-class distribution of live Black Box trees (0–15 cm DBH) at LMW; n = 975 in 2006–09, n = 1059 in 2009–12, n = 1004 in 2012–15, n = 854 in 2015–18.
3.6 Discussion

Due to the Black Box communities being higher on the floodplain, they experience less frequent inundation than communities lower on the floodplain. This factor also makes these areas inherently difficult to artificially inundate. As such, much of the Black Box population at LMW has not been inundated for a considerable time. While flooding in late 2016 inundated small areas of Black Box communities, the next largest flood was 14 years prior, with many areas of the population still not receiving water for even greater periods. While Black Box can use rainfall and ground water to maintain condition, inundation is generally required to cause germination (George 2004). As such, there has been little opportunity for Black Box to germinate across much of LMW for a considerable period of time. This is reflected in the comparatively low abundance of smaller sized trees.

Of note is the slow growth of Black Box across LMW. Some seedling-sized trees (0–3 cm DBH) have not increased substantially in size during nine years of monitoring (Pers. obs. D Wood). Closer investigation of individual trees may suggest this is a common pattern across much of the population. With reduced water availability, it may be that available energy is being used to expand root structure to maximise available moisture uptake or the trees have entered a state of dormancy.

Applying environmental water to benefit Black Box is difficult across much of LMW. A strategic approach may be required to target small areas of Black Box that are either in very poor health and/or are easier to inundate. This may facilitate the establishment of small refuges of Black Box in optimal condition. These refuges may then provide a seed bank that contributes to surrounding areas when environmental conditions are more favourable.

The target of more than 80% of Black Box with a crown condition score of ≥ 4 is currently being met. The minimum threshold relating to the population status index is not being met. Overall, the ecological objective relating to sustainable populations of Black Box at LMW icon site is only partially being met.
4 Wetland vegetation communities

Author: Louise Romanin

4.1 Introduction

Wetlands of the Murray–Darling Basin can be permanent, temporary or ephemeral bodies of water to various depths. During overbank flooding, wetlands become connected to river systems 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 river floodplain systems. 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 river floodplain habitat 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 (TLM) 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 Lindsay–Mulcra–Wallpolla (LMW) icon site has been undertaken since 2007–08. This chapter reports on the findings of these surveys over the last eleven years.

4.2 Ecological objectives

The vision for the LMW icon site is:

To maintain and restore a mosaic of healthy floodplain communities across Lindsay, Mulcra and Wallpolla Islands, which will ensure that indigenous plant and animal species and communities survive and flourish throughout the site (MDBA 2012b).

The overarching ecological objective for vegetation at LMW icon site is:

Increase the diversity, extent and abundance of wetland vegetation (MDBA 2012b).

The adopted objective for vegetation at LMW icon site is:

Restore diversity, extent and abundance of wetland aquatic vegetation (Robinson 2014b).

This chapter reports on the established vision statement and overarching ecological objective for wetlands at LMW. Following the detailed methods described for wetland vegetation in The Living Murray: Condition Monitoring Program design for the LMW, Mulcra and Wallpolla Islands (Huntley et al. 2016a), this chapter will report on the findings of research designed to:

- assess water responsive species richness and abundance against a point of reference
- assess the condition of the whole icon site using species richness and abundance scores
- examine the presence or absence of aquatic vegetation in wetlands.

4.3 Methods

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 LMW. Ten wetlands have been surveyed annually since 2007–08 with two exceptions: 1) in 2010–11 Upper Mullaroo Wetland Complex, Webster’s Lagoon, and Wetland 33 were inaccessible due to extensive flooding, and therefore not surveyed; 2) in 2014–15, no surveys were undertaken due to changes in program funding. Comprehensive detail on wetland vegetation assessments are provided in the Condition Monitoring Program design for the LMW icon site (Huntley et al. 2016a)
The total number of wetlands surveyed at LMW has risen to 12 following establishment of a new site at Lake Wallawalla in 2009-10, and the re-assigning of Scotties Billabong to a different water regime class. Scotties Billabong was previously surveyed as an extension of a floodplain site (Lindsay Island Floodplain Site 2). It has since demonstrated the ability to retain water following inundation, suggesting it is more suited to the definition of a wetland than a floodplain. Data from 2007–08 to 2015–16 (excluding 2014–15) were collected using methods described in the floodplain chapter of this report (Section 5), though it is proposed that this site be transitioned to wetland data collection methods over the coming years. Since reporting in 2013–14, Scotties Billabong has been included in the wetland chapter for discussion.

Each year, surveys were conducted between December and March following the methods described in the Condition Monitoring Program design for LMW (Huntley et al. 2016a).

### 4.3.1 Wetland classifications

Due to the limited hydrological information available for LMW, it was recommended that wetlands be analysed as one category (i.e. ‘wetlands’), rather than split into water regime classes (WRC) based on flooding frequency as has been the case in previous monitoring years (Brown et al. 2015).

### 4.3.2 Field survey methods

Wetland vegetation survey procedures were based on those developed by Nicol and Weedon (2006). Vegetation was sampled at 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 quadrats and elevations at each individual wetland, refer to Section 3 in Part B of this report. Details of the survey methods used can be found in Section 6 of The Living Murray Condition Monitoring Program design for LMW (Huntley et al. 2016a).

It is possible that some plant species that occur at the LMW icon site were not captured in the sampling method. Quadrat size (the use of 15, 1 m x 1 m cells) was determined from 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 Chowilla and LMW Islands (which technically comprise the one icon site), the same sampling intensity has been adopted for LMW. However, given the size of the quadrats in comparison to the area surveyed, there may be some species with patchy distributions or low abundances that were not captured within the sampled quadrats. Also, the seasonality of plant life cycles means that some species may not have been present at the time of the survey.

### 4.3.3 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.rbgs.g.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 version). Where species are not recorded for Victoria, scientific and common names follow the Flora of New South Wales (published and online versions).

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 (Campbell & Nielsen 2014).

Plant species recorded in surveys at LMW were classified into functional groups (Table 4.1). Functional group classification for each species is provided 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 the 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 4.1** Functional groups used to classify plant species recorded during The Living Murray surveys at LMW icon site.

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

Point of reference assessment

As part of an adaptive management process, The Living Murray Program recently underwent refinements to develop site-specific ecological targets (a point of reference) that link back to the vision statement and ecological objectives (Robinson 2014a, b). As the program continues to evolve, further improvements have been made and the ecological targets set in Robinson (2014b) and Huntley et al. (2016a) have been refined in this report to be consistent across TLM icon sites in the Mallee Catchment Management Authority region (i.e. Hattah Lakes and LMW) for all understorey vegetation components (i.e. wetland and floodplain communities). For consistency, the ecological targets for wetland vegetation (Table 4.2) were developed using data from 2007–08 to 2015–16 and following the methods described and recommended in Huntley et al. (2016a).

Table 4.2 Ecological targets for wetland vegetation communities at the LMW icon site (developed following the methods described in Huntley et al. (2016a)).

<table>
<thead>
<tr>
<th>Water regime class</th>
<th>Index 1: water responsive species richness (80th percentile)</th>
<th>Index 2: water responsive species abundance (80th percentile)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wetlands</td>
<td>4.85</td>
<td>32.47</td>
</tr>
</tbody>
</table>

Using the indices in Table 4.2, wetland vegetation was considered to be in good condition when:

- water responsive species richness was at or above the 80th percentile (adapted from Huntley et al. (2016a))
- water responsive species abundance was at or above the 80th percentile (adapted from Huntley et al. (2016a)).

To calculate if water responsive species richness was in good condition for wetlands (adapted from Huntley et al. (2016a):

- all years of data were used, including only water responsive plant species (see Section 4.3.3)
- the total number of species were averaged across all quadrats for each transect in each year
- transects with water responsive species richness at or above the 80th percentile (Index 1 in Table 4.2) score = 1 (i.e. compliant) and transects with water responsive species below the point of reference score = 0 (i.e. non-compliant)
- the proportion of compliant transects were plotted in a graph.

The same steps (above) were applied to determine if water responsive species abundance was in good condition using the sum of abundance and Index 2 in Table 4.2.

Aquatic vegetation in wetlands

The objective that relates to aquatic vegetation (macrophytes) is not clearly defined for wetlands at LMW (Robinson 2014b). For this report, the objective has been addressed by examining the presence/absence of aquatic vegetation in wetlands. Graphs that display the proportion of functional group abundance data for each monitoring year were created in Microsoft Excel. For display purposes, functional groups A, Arf, Arp, Ate, Atl (for definitions see Table 4.1) were combined into one amphibious group ‘A’. Functional group ‘T’ was excluded from these graphs as it was not possible to determine if these species were drought tolerant (Tdr) or terrestrial damp (Tda) species.
### 4.4 Results

#### 4.4.1 Wetland inundation state

The historic and current inundation state of each wetland provided context for data analyses. Twelve wetlands were surveyed in 2017–18. When surveys were undertaken in 2017–18, three wetlands were inundated, two wetlands were drawing down and seven had completely dried (Table 4.3). All wetlands were previously inundated by natural flooding in 2010–11 and 2016; environmental flows were delivered to wetlands in the years listed in Table 4.3.

**Table 4.3** Hydrological state of wetlands surveyed at LMW in 2017–18. Key: intermittent-dry = all quadrats dry, but wetland held water less than two years ago and may still display a vegetation response to inundation; drawdown = less than half the quadrats were inundated at the time of the survey; inundated = equal to or more than half the quadrats were inundated at the time of the survey.

<table>
<thead>
<tr>
<th>Island</th>
<th>Wetland</th>
<th>Hydrological state 2017-18</th>
<th>Inundated (Environmental &amp; natural)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Bottom Island</td>
<td>Drawdown</td>
<td>2010, 2016^</td>
</tr>
<tr>
<td></td>
<td>Lake Wallawalla</td>
<td>Inundated</td>
<td>2010; 2012; 2015; 2016, 2017</td>
</tr>
<tr>
<td></td>
<td>Scotties Billabong (Lindsay floodplain very often)</td>
<td>Intermittent-dry</td>
<td>2005; 2006; 2008; 2009; 2010, 2016</td>
</tr>
<tr>
<td></td>
<td>Upper Lindsay</td>
<td>Intermittent-dry</td>
<td>5010, 2016</td>
</tr>
<tr>
<td></td>
<td>Wetland 33</td>
<td>Drawdown</td>
<td>2010, 2016</td>
</tr>
</tbody>
</table>

^While Bottom Island did not receive any direct environmental water via pumping or other delivery means, the lower elevations of a number of transects are influenced by changes to the height of Lindsay River, which is affected by the operation of Locks and Weirs 6 and 7.

#### 4.4.2 Data summary

A total of 108 species was recorded in 2017–18 across all wetlands at the LMW icon site. The three most abundant species were water responsive species; Common Sneezeweed (*Centipeda cunninghamii*), Spreading Nuthead (*Spaeroptorphyceae littoralis*), and Hairy Carpet-Weed (*Glinus lotoides*). There were 10 non-native species and 15 species were recorded that have conservational significance in Victoria. Red Copperburr (*Sclerolaena calcarata*), a dry adapted species listed as endangered in Victoria, was recorded for the first time in TLM wetland surveys. For more detailed information about which species were recorded and where, refer to Part B of this report.
4.4.3  **Point of reference assessment**

**Water responsive species richness**

Scores were calculated for water responsive species richness for all wetlands at LMW, using the ecological target of 4.85, Index 1 in Table 4.2 (Figure 4.1a). Water responsive species richness had decreased slightly since 2016–17 as five wetlands had completely dried since that time. Water responsive species richness tends to respond positively to drying wetlands but decreases once a wetland is completely dry (Brock & Casanova 1997). Fewer plant species are expected at inundated wetlands as the plant community tends to be limited to a few floating and/or submerged species. Water responsive species richness scores were highest in 2011–12, approximately twelve months following flooding in 2010–11. The second greatest scores were recorded in 2008–09, following environmental flows that were delivered to seven wetlands in 2008.

**Water responsive species abundance**

Scores were calculated for water responsive species abundance for all wetlands at LMW, using the ecological target of 32.47, Index 2, in Table 4.2 (Figure 4.1b). Unlike the results for species richness (above), species abundance in 2017–18 has decreased dramatically since 2016–17. Abundance scores this year have returned to levels typical of dry periods. Water responsive species abundance was greatest in 2008–09 following delivery of environmental flows and in 2011–12 following flooding in 2010–11.

![Figure 4.1](image)

**Figure 4.1** Proportion of transects (± SE) meeting the reference value for (a) water responsive species richness and (b) water responsive species abundance scores for wetlands at LMW. Scores are based on 80th percentile indices. Wetlands surveyed; n = 11 in 2007–08 and 2008–09; n = 12 in 2009–10, 2011–12 to 2017–18; n = 9 in 2010–11.
4.4.4 Aquatic vegetation in wetlands

Community composition was comprised of species from terrestrial dry, terrestrial damp and amphibious woody plants functional groups in most monitoring years (Figure 4.2). The proportion of damp responsive species had decreased during the 2017–18 survey period, following the drying of wetlands previously inundated in 2016–17. Many of the damp responsive species that had grown in wetlands were dead by the time this year’s surveys were undertaken (Pers. obs. L Romanin) (Figure 4.4). In 2010–11, the community predominantly consisted of floating species due to the level of inundation at the time surveys were undertaken. The proportion of water responsive species, amphibious and terrestrial damp, was greatest in years immediately following inundation (i.e. 2008–09 following environmental flows, 2011–12 and 2012–13 following flooding in 2010–11, and 2016–17 following flooding in November 2016).

Figure 4.2 Proportion of sum of abundance for each functional group across all wetlands for each monitoring year. Wetlands surveyed; \( n = 11 \) in 2007–08 and 2008–09; \( n = 12 \) in 2009–10, 2011–12 to 2017–18; \( n = 9 \) in 2010–11. Key: plant species functional groups; \( S = \) submerged, \( F = \) floating detached, \( A = \) aquatic and amphibious species, \( A_{tw} = \) amphibious woody plants, \( T_{da} = \) terrestrial damp species, \( T_{dr} = \) drought tolerant plants.

4.5 Discussion

Community composition was comprised of species from a suite of terrestrial and amphibious functional groups in all monitoring years. One survey year is an outlier, in 2010–11 the community predominantly consisted of floating species as sites were inundated when surveys were undertaken. Effects from this flood (e.g. the presence of terrestrial damp and amphibious species) were evident for two years following flooding (e.g. 2011–12 and 2012–13 surveys). Since then, an increase in drought tolerant species has occurred as the inter-flood dry period has increased.

Widespread flooding in November 2016 inundated all wetlands at LMW. Wetlands were in various wet/dry states during 2017–18 surveys (e.g. inundated, intermittent-dry or drawing down). Usually, drying wetlands provide a wet/dry ecotone that is particularly high in species richness (Brock & Casanova 1997). However, during this year’s surveys we found decreased species richness and abundance as well as an increase in the proportion of drought tolerant species compared to 2016–17 surveys. During the summer of 2017–18, rainfall was much lower than long term average and daily maximum temperatures were higher than average (BOM 2018). These hot, dry conditions may have hastened the drying of wetlands inundated in 2016, making the sites unsuitable for water responsive species (Figure 4.3 & Figure 4.4).
Summary

Key points from condition monitoring of wetland vegetation at LMW in 2017–18 are:

- Most wetlands had dried after being inundated in November 2016.
- In response to drying conditions, there has been a decrease in the proportion of terrestrial damp and amphibious vegetation and an increase in the proportion of drought tolerant plant species.

4.5.1 Progress towards ecological objectives

Wetland vegetation communities at LMW were reverting back to those characteristic of a dry phase after flood waters from 2016 had mostly receded. During 2017–18 surveys, most wetlands had dried but three retained water, supporting a mosaic of habitats across LMW. Terrestrial damp and amphibious plant species richness was higher than in dry years but the abundance of these plants was low across the LMW icon site. Cycles of wetting and drying are occurring at LMW. Therefore, the overarching vision for wetlands at LMW to maintain a mosaic of healthy wetland communities is being met in 2017–18.
4.6 Recommendations

Wetland indices were refined in this report based on information in Brown et al. (2015) and Huntley et al. (2016a). For consistency, these targets should be updated in The Living Murray Condition Monitoring Program design for LMW Islands (Huntley et al. 2016).

Now that reference values have been trialled for each icon site, it is suggested that a workshop be held to discuss outcomes of this approach. This should include scientists, field work researchers and contractors, and TLM icon site managers. This may assist in addressing queries such as; what is the best way to evaluate low species richness and abundance scores during inundation?

As part of an intervention monitoring program funded by the Mallee CMA, additional wetland sites were established at Mulcra Island to specifically capture a response to environmental watering. Four sites were established and surveyed in 2011 (prior to delivery of environmental water) and 2013 (following delivery of environmental water). Although data were collected, they have not yet been analysed. We recommend analysis of these data, as they could provide valuable insight into the effect of environmental water on wetland vegetation at Mulcra Island.

A combination of environmental flows and unregulated flooding appears to be supporting water responsive rare plants. There is limited information about these species, largely because of their ephemeral nature. Threatened species recovery plans and action plans are about determining how many populations exist and mapping 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 such as 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.
5  Floodplain vegetation communities

AUTHOR: LOUISE ROMAIN

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. 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 (TLM) 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 Lindsay–Mulcra–Wallpolla (LMW) icon site has been undertaken since 2007–08. This chapter reports on the findings of these surveys over the last decade.

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.

This chapter reports on the established vision statement and overarching ecological objective for floodplain vegetation at LMW icon site. Following the detailed methods described for floodplain vegetation in The Living Murray Condition Monitoring Program design for LMW (Huntley et al. 2016a), this chapter will report on research designed to:

- assess species richness and species abundance against a point of reference
- assess the condition of the whole icon site using species richness and abundance scores
- analyse changes in vegetation community composition over time.

5.3  Methods

There are six floodplain understorey monitoring locations established under The Living Murray Program at the LMW icon site, with each island having two locations (Lindsay = L1 and L2, Mulcra = M1 and M2, and Wallpolla = W1 and W2). Locations were established in 2007–08, each of which consists of three Flood return frequencies (FRFs) (i.e. one often, one sometimes and one rarely-flooded site per FRF = 18 sites) (Table 5.1). Comprehensive detail on floodplain vegetation assessments is provided in the Condition Monitoring Program design for the LMW icon site (Huntley et al. 2016a).

Surveys have been undertaken annually at these 18 sites since 2007–08. The exceptions being in 2010–11 when only 15 sites were surveyed as flooding prevented access to some sites, and in 2014–15 when no data were collected due to changes in program funding. Each year, surveys were conducted following the methods described in The Living Murray Condition Monitoring Program design for LMW icon site (Huntley et al. 2016a).
Table 5.1 Flood return frequencies (FRFs), floodplain elevation, commence-to-flow level and associated floodplain site names for The Living Murray condition monitoring program at the LMW icon site. The FRFs were determined using commence-to-flow data. Key: site names, L = Lindsay Island, M = Mulcra Island, W = Wallpolla Island.

<table>
<thead>
<tr>
<th>Flood Return Frequency</th>
<th>Floodplain elevation</th>
<th>Commence to Flow</th>
<th>Site names</th>
</tr>
</thead>
<tbody>
<tr>
<td>Often</td>
<td>Lower floodplain</td>
<td>35 000–60 000 ML day⁻¹</td>
<td>L1A; L2A; M1A; M2A; W1A; W2A</td>
</tr>
<tr>
<td>Sometimes</td>
<td>Mid floodplain</td>
<td>60 000–100 000 ML day⁻¹</td>
<td>L1B; L2B; M1B; M2B; W1B; W2B</td>
</tr>
<tr>
<td>Rarely</td>
<td>Higher floodplain</td>
<td>&gt; 100 000 ML day⁻¹</td>
<td>L1C; L2C; M1C; M2C; W1C; W2C</td>
</tr>
</tbody>
</table>

5.3.1 Field survey methods

At each location in each FRF, there are four permanently established quadrats, spaced 50 m apart and each consisting of 15 x 1 m x 1 m cells. Floodplain vegetation surveys follow the methods described in section 4.3.2. The methods to identify plant species and the use of plant functional group are described in section 4.3.3.

5.3.2 Data analysis

Point of reference assessment

As part of an adaptive management process, The Living Murray Program recently underwent refinements to develop site-specific ecological targets that link back to the vision statement and ecological objectives (Robinson 2014a, b). As the program continues to evolve, further improvements have been made, and the ecological targets set in Robinson (2014b) and Huntley et al. (2016a) have been refined in this report to be consistent across the Mallee CMA TLM icon sites for all understorey vegetation components (i.e. wetland and floodplain communities). For consistency, the ecological targets for floodplain vegetation (Table 5.2) were developed using data from 2007–08 to 2015–16 and following the methods described and recommended in Huntley et al. (2016a).

Table 5.2 Ecological targets for floodplain understorey vegetation communities at the LMW icon site (developed using data from 2007–08 to 2015–16 and following the methods described in Huntley et al. (2016a).

<table>
<thead>
<tr>
<th>Flood Return Frequency</th>
<th>Index 1: water responsive species richness (80th percentile)</th>
<th>Index 2: water responsive species abundance (80th percentile)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lower floodplain</td>
<td>5.45</td>
<td>25.3</td>
</tr>
<tr>
<td>Mid floodplain</td>
<td>3.25</td>
<td>11.45</td>
</tr>
<tr>
<td>Higher floodplain</td>
<td>1.5</td>
<td>3.7</td>
</tr>
</tbody>
</table>

Using the indices in Table 5.2, floodplain vegetation communities were deemed to be in good condition when:

- water responsive species richness in a FRF is at or above the 80th percentile (adapted from Huntley et al. (2016a)).
- water responsive species abundance in a FRF is at or above the 80th percentile (adapted from Huntley et al. (2016a)).
To calculate if water responsive species richness was in ‘good’ condition for floodplains (adapted from Huntley et al. (2016a)):

- all years of data were used, including only water responsive plant species (see Section 4.3.4)
- the total number of species were averaged across all quadrats for each site in each year
- for each FRF, sites with water responsive species richness at or above the 80th percentile (Index 1 in Table 5.2) score = 1 (i.e. compliant) and sites with water responsive species below the point of reference score = 0 (i.e. non-compliant)
- the proportion of compliant quadrats were plotted on a graph for each FRF.

The same steps (above) were applied to determine if water responsive species abundance was in ‘good’ condition for each FRF using the sum of abundance of water responsive plant species.

A Whole-Of-Icon site score was determined for water responsive species richness using the methods described in Brown et al. (2015). The calculation of this score requires that FRF scores be weighted according to the relative floodplain area within each FRF at LMW (Brown et al. 2015). The statistical weighting follows the process detailed in (Sutherland 2006) and a worked example, showing equations, is presented in (Brown et al. 2015). These Whole-Of-Icon site scores were calculated for each survey year since 2007-08 (excluding 2014-15). These values were plotted as a time series to examine the effect environmental watering has had on the richness and abundance of water responsive species at an icon site scale.

**Plant functional groups**

The use of plant 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 help demonstrate the impact of flood inundation on community composition. Graphs that display the proportion of functional group abundance data for each monitoring year, in each FRF, were created in Microsoft Excel. For display purposes, functional groups A, Arf, Arp, Ate, Atl (for definitions see Table 4.1 in Section 4.3.3) were combined into one amphibious functional group ‘A’. Functional group ‘T’ was excluded from these graphs as it was not possible to determine if these species were drought tolerant (Tdr) or terrestrial damp species (Tda).

5.4 Results

5.4.1 Floodplain inundation state

The historic and current inundation state of each floodplain site provided context for data analysis (Table 5.3). None of the sites were inundated during this monitoring period. Flooding in 2016 inundated the majority of floodplain sites, including two of eight long-dry sites, which were previously inundated in 1993–94. Five sites remain long-dry and have not been inundated for ~23 years (i.e. since 1993–94).
Table 5.3 Hydrological state of each floodplain site surveyed at LMW in 2017–18. Key: RRG = River Red Gum overstorey, BB = Black Box overstorey; Intermittent-dry = dry during survey, but held water less than two years ago and may still show some vegetation response to inundation, Long-dry = dry for at least the last two monitoring years.

<table>
<thead>
<tr>
<th>Flood Return Frequency</th>
<th>Site</th>
<th>Vegetation community</th>
<th>Hydrological state in 2017–18 surveys</th>
<th>Previously inundated</th>
</tr>
</thead>
<tbody>
<tr>
<td>Lower floodplain</td>
<td>L1A</td>
<td>RRG</td>
<td>Intermittent-dry</td>
<td>2010–11, 2016</td>
</tr>
<tr>
<td>Lower floodplain</td>
<td>M1A</td>
<td>RRG</td>
<td>Intermittent-dry</td>
<td>Nov 2016</td>
</tr>
<tr>
<td>Lower floodplain</td>
<td>W1A</td>
<td>RRG</td>
<td>Intermittent-dry</td>
<td>2010–11, 2016</td>
</tr>
<tr>
<td>Mid floodplain</td>
<td>M1B</td>
<td>BB</td>
<td>Intermittent-dry</td>
<td>2010–11, 2016</td>
</tr>
<tr>
<td>Mid floodplain</td>
<td>W1B</td>
<td>BB</td>
<td>Intermittent-dry</td>
<td>2010–11, 2016</td>
</tr>
<tr>
<td>Mid floodplain</td>
<td>W2B</td>
<td>BB</td>
<td>Long-dry</td>
<td>~1993–94</td>
</tr>
<tr>
<td>Higher floodplain</td>
<td>L2C</td>
<td>BB</td>
<td>Long-dry</td>
<td>~1993–94</td>
</tr>
<tr>
<td>Higher floodplain</td>
<td>M1C</td>
<td>BB</td>
<td>Intermittent-dry</td>
<td>~1993–94, 2016 (partial)</td>
</tr>
<tr>
<td>Higher floodplain</td>
<td>M2C</td>
<td>BB</td>
<td>Long-dry</td>
<td>~1993–94</td>
</tr>
<tr>
<td>Higher floodplain</td>
<td>W1C</td>
<td>BB</td>
<td>Long-dry</td>
<td>~1993–94</td>
</tr>
</tbody>
</table>

5.4.2 Data summary

A total of 83 plant species was recorded in 2017–18 on the floodplains at LMW. The most abundant species was the drought tolerant Ruby Saltbush (*Enchylaena tomentosa* var. *tomentosa*). The damp responding species Caustic Weed (*Euphorbia dallachyana*) and Common Sneezeweed (*Centipeda cunninghamii*) were also common. There were six non-native species recorded and 10 species that have conservational significance in Victoria. Branching Groundsel (*Senecio cunninghamii* var *cunninghamii*) is listed as rare in Victoria but was reasonably common this year. For more detailed information about which species were recorded and where, refer to Part B of this report.

5.4.3 Point of reference assessment

Water responsive species richness

Scores were calculated for water responsive species richness for each FRF using Index 1 of the ecological targets in Table 5.2 (Figure 5.1a). At often and sometimes-flooded sites, water responsive species richness had decreased from the peak caused by flooding in 2016. Sometimes-flooded sites still had elevated scores compared to non-watered years. Water responsive species richness for rarely-flooded sites had decreased compared to 2016–17.
**Water responsive species abundance**

Scores were calculated for water responsive species abundance for each FRF using Index 2 of the ecological targets in Table 5.2 (Figure 5.1b). Water responsive species abundance had decreased at each FRF category compared to 2016–17, but was elevated compared to dry years at often and sometimes-flooded sites. Water responsive species abundance at rarely-flooded sites was the same as that found in dry years.

![Graph](image)

**Figure 5.1** Proportion of transects (± SE) meeting the reference value for (a) water responsive species richness scores and (b) water responsive species abundance scores for floodplains at LMW (n = 18 sites in all years except 2010–11 where n = 15). No surveys were undertaken in 2014–15. Scores were based on 80th percentile indices, specific to each flood return frequency (i.e. often-, sometimes-, and rarely-flooded).
5.4.5 Whole-of-icon site scores

A Whole-of-icon site score was determined for both water responsive species richness (Figure 5.2) and abundance (Figure 5.3) for each year that monitoring has occurred. Calculations were based on the number of sites surveyed in 2017–18 that were at or above the point of reference listed in Table 5.2 for each FRF. The weighted icon site score for species richness was 0.476 (±0.328) and the weighted icon site score for species abundance was 0.25 (±0.284) during the 2017–18 surveys. Both of these values were lower than the values calculated for 2016–17.

When plotted as a time series, fluctuations are evident in species richness and abundance in response to natural flooding or the delivery of environmental water (Figure 5.2a & Figure 5.3a). Species richness first declined in response to drought-breaking natural floods in 2010-11 which inundated the LMW Icon site. Often- and sometimes-flooded sites were inundated at this time and were, therefore, low in species richness. At the same time rarely-flooded sites had naturally low water responsive species richness due to being predominantly dry. The first peak in species richness is seen in 2011–12 while flooding continued at the icon site. A peak of even larger magnitude was caused by flooding across the LMW Icon site in November of 2016. Although there are peaks in richness across the icon site, these are not significant differences as there is high variability within flooded and non-flooded periods.

Much the same pattern can be seen in Whole-of-icon site species abundance scores as in richness scores. The notable difference is that species abundance is very low in the environmental watering category (Figure 5.3b). This was possibly due to extensive inundation; patterns in response to environmental watering were not detected as no monitoring was funded in 2014–15.
Figure 5.2 (a) Whole-of-icon site scores (± 95% CI for two sampled comparisons with normally distributed error variance) for water responsive species richness at LMW floodplains. The total area in each flood return frequency (FRF) was used to weight the average strata scores for each FRF into an overall icon site mean score. (b) The mean Whole-of-Icon site score (± SE) of each flooding period as indicated (a) above; non-flooded (white) n = 6, natural flooding (green) n = 3, environmental water (e-water) (blue) n = 1.
5.4.6 Plant functional groups

Often-flooded communities were made up of species from an array of functional groups in almost all monitoring years (Figure 5.4a). Following flooding in 2016, species from an array of amphibious and terrestrial damp functional groups emerged. In response to the recession of flood waters during this survey year the proportion of drought tolerant and (curiously) aquatic species increased, but total abundance of all functional groups dropped drastically. Floating species dominated the plant
Figure 5.4 Proportion of sum of abundance for each functional group in (a) often-, (b) sometimes- and (c) rarely-flooded categories. (n = 18 sites in all years except 2010–11 where n = 15). No surveys were undertaken in 2014–15. Key: plant species functional groups; S = submerged, F = floating detached, A = aquatic and amphibious species, Atw = amphibious woody plants, Tda = terrestrial damp species, Tdr = drought tolerant plants.
community in 2010–11 as sites were inundated when surveys were undertaken. Since then, the proportion of drought tolerant species has increased as the inter-flood dry period has lengthened. The proportion of drought tolerant species reached a peak in 2015–16.

There was a greater proportion of drought tolerant species in all monitoring years at sometimes-flooded sites (Figure 4b) compared with often-flooded sites (Figure 4a). The proportion of water responsive and amphibious species had declined this monitoring year compared to 2016–17.

Rarely-flooded sites were dominated by drought tolerant species in all monitoring years (Figure 4c).

5.5 Discussion

Understorey vegetation communities in often- and sometimes-flooded sites still retained some water responsive species, albeit at lower abundances and proportions than during the last survey season. Often- and sometimes-flooded sites were made up of species from a suite of terrestrial and amphibious functional groups in 2017–18. Communities in rarely-flooded sites remained dominated by drought tolerant species. The majority of rarely-flooded sites remained long-dry and have not been inundated for up to 24 years (i.e. since 1993–94).

A total of 83 plant species were recorded in 2017–18, the most abundant of which was the drought tolerant Ruby Saltbush (Enchylaena tomentosa var. tomentosa). For the first time since monitoring began, the vulnerable, drought-tolerant species Poverty-bush (Sclerolaena intricata), was found at LMW floodplains.

Overall, results indicate that the water responsive plant community at LMW at often- and sometimes-flooded elevations still benefitted from flooding that occurred in 2016.

5.5.1 Often-flooded

Community composition in often-flooded sites has changed over time. In 2010–11, the lower floodplain was dominated by floating plant species, due to the extent of inundation at the time surveys were undertaken. Since then, the drought tolerant proportion of the community continuously increased as the inter-flood dry period lengthened. In 2016–17, there was a substantial increase in the terrestrial damp and amphibious proportion of the community in response to widespread inundation in November 2016. During the subsequent year, often-flooded sites have dried, resulting in the proportion of wet responding species decreasing.

Often-flooded sites have shown strong responses to flooding. The greatest water responsive species richness and abundance scores were recorded in 2011–12 (12 months following flooding in 2010–11) and in 2016–17 (~2–3 months following flooding in November 2016). These results are in line with expectations, as drying wetlands and their surrounding floodplains provide a wet/dry ecotone that is particularly high in species richness (Brock & Casanova 1997).

5.5.2 Sometimes-flooded

Changes in community composition at sometimes-flooded sites were similar to those in often-flooded sites, but with a typically higher proportion of drought tolerant species in all monitoring years. In 2017–18, community composition was made up of species from a suite of terrestrial and amphibious functional groups. The measured values for these sites had decreased since 2016–17 when flooding affected most areas. As with often-flooded sites, sometimes-flooded sites recorded the highest water responsive species richness and abundance scores in 2011–12 and 2016–17, following recession of floodwater.
5.5.3 Rarely-flooded

Rarely-flooded sites were dominated by drought tolerant species in all monitoring years with little shift in community composition over time. Two of the six sites, previously flooded in 1993–94, were partially and shallowly inundated in November 2016. At these sites, drought tolerant species benefited from shallow inundation and very little response from terrestrial damp or amphibious species was seen. This lack of response could be due to the rapid recession of floodwater, or the shallow depth, or could be a reflection of the condition of soil seedbank stores after a long inter-flood dry period. The remaining higher floodplain sites are long-dry and were last inundated ~23 years ago (in 1993–94).

Water responsive species richness and abundance scores at rarely-flooded sites had dropped back to those seen prior to flooding in 2016. The highest water responsive species scores were recorded in 2016–17 and 2007–08. These results were unexpected given surveys in 2007–08 were undertaken during severe drought conditions. As explained in Brown et al. (2016), the only available data to set these targets comes from a period when these sites were long-dry. Therefore, few water responsive species were present in this data set, potentially making the ecological target easier to achieve and less reflective of response to inundation (e.g. higher floodplain sites only need to record two water responsive species to meet the target for Index 1 in Brown et al. (2016). This may explain why scores for higher floodplain sites were comparatively greater than lower and mid floodplain sites, even though the higher floodplain was long-dry (e.g. results in 2008–09 and 2015–16 in Figure 5.1).

5.5.4 Whole-of-icon site scores

This is the first time the Whole-of-icon site scores have been calculated for the entire time series of TLM monitoring at LMW floodplains. We have found the score to be sensitive to flooding. Increases in richness and abundance of water responsive species were detected following flooding periods. It appears that there is a lag between flooding and an increase in species richness, with receding floodwaters providing ideal niches for water-responding terrestrial species to complete their life cycles. We found a minimal response to environmental watering at LMW but this could be attributed to the lag described above which was not detected due to a year of missed surveying.

No trend of increased species richness or abundance over the icon site was detected.

5.5.5 Progress towards ecological objectives

The water responsive plant community benefited from flooding in November 2016 at the often- and sometimes-flooded sites. The rarely-flooded areas are predominantly long-dry and remain dominated by drought tolerant species. Therefore, the overarching vision to maintain healthy floodplain communities at LMW is partially being met.

5.5.6 Summary

Key points from condition monitoring of floodplain understorey vegetation at LMW in 2017–18 are:

- Water responsive plant species at often- and sometimes-flooded sites of the LMW floodplain show some continued benefit after being flooded in November 2016. Overall, however, water responsive species richness and abundance has decreased since 2016–17.

Vegetation community composition has changed little across all monitoring years and remains dominated by drought tolerant species. Though two sites were partially and shallowly inundated in November 2016, the majority of rarely-flooded sites remain long-dry and were last inundated ~23 years ago (in 1993–94).
5.6 Recommendations

As recommended in Brown et al. (2016), now that references have been trialled for each icon site, a workshop should be held to include scientists, field work researchers/contractors and TLM icon site managers to discuss outcomes of this approach. This may assist in addressing the following sorts of queries:

- Why do sites on the higher floodplain at LMW score better than often and sometimes-flooded sites in some years? Is this because the reference index uses data from a period of minimal or no inundation of the higher floodplain sites?
- Ecological targets have been set using data from periods when sites were predominantly long-dry as this is the only data available (Index 1 & 2 in Table 5.2). It is possible that this has made the targets easier to reach and less reflective of response to inundation (Brown et al. 2016).

A method to determine differences in the Whole-of-icon site scores between surveyed years should be developed. Additionally, a minimum target score may be considered in order to assess whether ecological targets are being met. The methods described in Richardson (2014) are missing important details required to do this. We recommend that resources be allocated to improving on this statistical method.

The lack of response to inundation at higher elevations of the floodplain could be as a result of the type of inundation (e.g. shallow inundation and rapid recession) or could be an indication of the condition of the soil seedbank. The drying process in arid floodplains is important for understorey vegetation communities to enable plant species to have time to complete their life-cycle stages, as well as for chemical and nutrient cycling processes (Boulton et al. 2014). However, intermittent inundation is required to sustain aquatic and amphibious floodplain communities. To improve our understanding of the condition of soil seedbanks, it would be beneficial to undertake soil germination trials, particularly in long-dry areas of the floodplain.

As part of an intervention monitoring program funded by the Mallee CMA, additional floodplain sites were established at Mulcra Island to specifically capture a response to environmental watering. Nine sites — three lower, three mid and three higher on the floodplain — were established and surveyed in 2011 and 2013, (prior to and following delivery of environmental water, respectively). Although condition monitoring sites were not influenced by environmental flows, it is likely that some of the intervention sites were inundated by flows delivered to Mulcra Island in 2013. Although data were collected, they have not yet been analysed. Analysis is recommended as this data could provide valuable insight into the effect of environmental water on floodplain understorey vegetation condition.
6 Lignum

AUTHOR: DAVID WOOD

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). It is a native shrub comprising of a tangled mass of normally leafless stems growing up to 3 m tall and 3 m wide. Communities are made up of separate male and female plants that can form dense thickets, dominating large areas of floodplain throughout the Murray–Darling Basin (Cunningham et al. 1992; Sainty & Jacobs 1981).

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). During extended dry periods, above-ground biomass of Lignum may appear lifeless and dead (dry, brown leafless stems), but may remain viable as underground rootstock. Within 2-4 weeks of heavy rainfall or flooding, Lignum rootstock may regenerate from dormancy, producing a green flush of shoots, leaves and flowers (Craig et al. 1991; Jensen 2008). Regeneration, however, can be highly variable, with the likelihood of successful regeneration varying among locations and diminishing with increased length of dormancy (Freestone et al. 2017). Ideally, Lignum requires flooding every 3 to 10 years (possibly more frequently in saline soils) (Craig et al. 1991) because once dormant, the maximum critical time period during which successful regeneration can occur, may be exceeded.

Monitoring of Lignum at Lindsay–Mulcra–Wallpolla Islands (LMW) as part of The Living Murray (TLM) condition monitoring first occurred during 2007. Monitoring has occurred annually, with the exception of 2014–15 as the TLM Program did not run. Recently, a review of the TLM Program was undertaken, with recommendations suggesting changes to the survey design, data collection method and new analyses (Brown et al. 2015; Huntley et al. 2016a; Robinson 2014b). These recommendations were first incorporated into the 2016–17 monitoring program; however, flooding during surveys meant all sites could not be assessed. This sampling period (2017–18) is the first year that all sites could be assessed using the new method recommended during the review process.

6.2 Ecological objectives

Ecological objectives for LMW Islands have been in refinement since interim objectives were first developed by the Ministerial Council in 2003 (MDBMC 2003). The most recent version of the ecological objective for Lignum is based on an understanding of Lignum responses to environmental conditions learned through monitoring, evaluation, research, modelling and consultation activities over ten years (MDBA 2012b). The ecological objective for Lignum at LMW is (MDBA 2012b):

*Improve condition and increase extent to sustain species assemblages and processes typical of Lignum communities.*

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

*Improve condition of Lignum communities*
6.3 Method

At LMW, Lignum was assessed across 26 sites, from 3 different strata: Lignum Shrubland, Lignum Woodland and Lignum Swamp. Surveys were undertaken annually at established sites consisting of 20 m X 20 m quadrats. Specifically, condition of all Lignum plants within each quadrat was assessed using the Lignum Condition Index (LCI) (Table 6.1). Additionally, each plant is assessed for leaf and flower abundance and the sex determined where possible. Lignum surveys for the 2017–18 monitoring period occurred during spring 2017.

Table 6.1 The Lignum Condition Index (LCI) used to assess Lignum plant condition. Adapted from Scholz et al. (2007a).

<table>
<thead>
<tr>
<th>% Viable</th>
<th>Score</th>
<th>Colour</th>
<th>Score</th>
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<tbody>
<tr>
<td>&gt; 95</td>
<td>6</td>
<td>All green</td>
<td>5</td>
</tr>
<tr>
<td>75 ≤ 95</td>
<td>5</td>
<td>Mainly green</td>
<td>4</td>
</tr>
<tr>
<td>50 ≤ 75</td>
<td>4</td>
<td>Half green, half yellow/brown</td>
<td>3</td>
</tr>
<tr>
<td>25 ≤ 50</td>
<td>3</td>
<td>Mainly yellow/brown</td>
<td>2</td>
</tr>
<tr>
<td>5 ≤ 25</td>
<td>2</td>
<td>All yellow/brown</td>
<td>1</td>
</tr>
<tr>
<td>0 ≤ 5</td>
<td>1</td>
<td>No viable stems</td>
<td>0</td>
</tr>
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<td>0</td>
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<td></td>
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</tr>
</tbody>
</table>

Comprehensive detail on Lignum condition monitoring methods is available in the Condition Monitoring Program design for the LMW icon site (Huntley et al. 2016a).

6.4 Indices and points of reference

The target developed for Lignum condition at LMW is:

more than 70% of Lignum plants at LMW have a LCI score of ≥4

At a strata level, Lignum Condition Index scores of plants were pooled and the percent of Lignum plants with an LCI of ≥4 was calculated. Each stratum was then assessed as being either ‘compliant’ or ‘non-compliant’ where:

- compliant = 1, if ≥70% plants had LCI score of ≥4,
- non-compliant = 0, if <70% plants had LCI score ≥4

At the icon site level, the Lignum Condition Index scores of all plants were pooled and the percent of plants with an LCI of ≥4 was calculated. If more than 70% of the plants had LCI scores ≥4 the site was deemed to be in good condition and to have attained the site-specific target.

To assist with reporting at an icon site level (as opposed to individual strata), indices for Lignum were determined by calculating each site as compliant or non-compliant. The percent of compliant sites (i.e. sites with ≥70% plants that are at or above the reference condition) was used to determine the icon site Condition Index. Annual changes to this Index indicate changes to Lignum condition, whereby a change of 0.3 between years will indicate significant changes in Lignum condition (Robinson 2014b).
6.5 Results

Monitoring of Lignum during 2017–18 revealed an improvement in condition for Lignum Swamp and Woodland strata as indicated by the increased frequency of plants with a score ≥4 from 2016–17 (Figure 6.1 b & c). However, this increase in condition resulted in only Lignum Woodland achieving the target of ≥70% plants with a score ≥4. This target was not achieved for either Lignum Swamp or Lignum Shrubland and there has been a decline in the frequency of Shrubland Lignum plants with a score ≥4 in the past year (Figure 6.1 a). As such, Lignum Woodland is the only strata to achieve compliance this monitoring period. This is the first time a strata has achieved the target value using the new, refined method.

![Figure 6.1](image)

Figure 6.1 Mean frequency (± SE) of Lignum plants with a LCI score of ≥4 across two years of sampling, for a) Lignum Shrubland (2016 n= 5, 2017 n=7), b) Lignum Swamp (2016 n=4, 2017 n=5), c) Lignum Woodland (2016 n=7, 2017 n=14) and d) icon site level (2016 n=16, 2017 n=26)(target value = 70%).

At the icon site scale, Lignum condition did not reach the target of ≥70% plants with a LCI score ≥4 (Figure 6.1 d). However, there was an improvement in condition, as indicated by an increase in the mean frequency of plants with a LCI score ≥4. As such, the icon site Condition Index increased from 0.375 in 2016–17 to 0.667 in 2017–18. This value represents the proportion of sites that exceed the target value.
6.6 Discussion

Lignum condition at LMW is highly variable, reflecting the different landscapes and hydrology across the icon site. Extensive flooding during late 2016 would have been the greatest driver of Lignum condition in the past two years, with 24 of the 26 sites inundated at this time. Lignum responds rapidly to inundation by producing a proliferation of green shoots, and these were still evident during 2017–18 monitoring, 10 months post flooding. Overall, Lignum condition was assessed as better in 2017–18, following inundation, as opposed to 2016–17.

A marked difference between the two surveys was that much of the floodplain was still deeply inundated during 2016–17. As such, many Lignum plants would not have been included in assessments as they were underwater at the time and not visible. This would be particularly apparent for smaller and dead plants (where dead stems collapse, ending up prone on the ground). During 2017–18, there was no inundation of the floodplain and all plants could be assessed, resulting in a greater number of dead plants being recorded.

The improvement in Lignum condition between the two sampling periods may also be an artefact of location. Sites not assessed during 2016–17 were more deeply submerged (lower on the floodplain) than those that were assessed. As such, these areas may have historically received more frequent inundation, resulting in the maintenance of these Lignum communities in better condition than communities higher on the floodplain, which were flooded less often.

In addition to inundation resulting in improved condition, germination of seedlings (or proliferation of vegetative growth) was also evident in some communities. Evidence of seedling germination in the past has been sparse so this represents a positive direction for Lignum communities by way of addressing ongoing mortality. Seedlings generally scored highly during assessment as the plant usually consists of a single green shoot, thus having high viability and colour scores.

With the recent change to methods, the way the data is collected has changed. At this time, all historic data cannot be viewed and interpreted in conjunction with the new data. For the past two monitoring periods, both methods have been employed in the field, with the intention of using this data to assess the compatibility between methods. Now that overlapping data exists, it is recommended that the compatibility of the methods be tested against each other.

As the methods have changed, there is no historic data with which to develop a minimum icon site Condition. As we collect more years of information and gain a better picture of Lignum condition at LMW, it is recommended that a minimum icon site Condition is developed. If testing of compatibility indicates that data from the old method is compatible with the new method this will provide a wealth of data toward determining an icon site minimum condition earlier than otherwise anticipated.

While Lignum condition showed some improvement in 2017–18, only a single strata, Lignum Woodland, met the target of ≥70% plants having a LCI score of ≥4 (compliant). At an icon site level there was an improvement in the icon site Condition Index from 0.375 in 2016–17 to 0.667 in 2017–18.
7 Cumbungi

AUTHOR: DAVID WOOD

7.1 Introduction

Cumbungi (Typha spp.) is a common macrophyte, native to the Murray-Darling basin (Scholz et al. 2007b). It comprises of two species; Narrow-leaf Cumbungi (Typha domingensis Pers.) and Broadleaf Cumbungi (T. orientalis Presl.), which are not distinguished between for the sake of monitoring due to their identical morphology, physiology and habitat preferences. Mature Cumbungi plants are erect, rhizomatous perennials that grow to 4 m in height and occupy riparian and wetland habitats (Sainty & Jacobs 1981).

Cumbungi is well adapted to rapid invasion and colonisation of wetland and riparian habitat due to its specific biology and ecology (Nicol & Ganf 2000; Roberts & Marston 2000). The altered flow regimes caused by river regulation favour the proliferation of Cumbungi (Roberts & Marston 2000). In some instances, it can form monospecific cultures that choke waterways and wetlands. This reduces aquatic flora diversity, reduces hydraulic capacity and can prevent fish passage (Roberts & Marston 2000; Roberts & Wylls 1992). Under the correct conditions, Cumbungi plays an important ecological role in providing habitat for fish, macroinvertebrates, zooplankton and water-birds (Ogden 2000; Roberts & Marston 2000).

Cumbungi has previously been recorded in all reaches at Lindsay-Mulcra-Wallpolla (LMW) icon site. Toward the end of the millennium drought (2001–2009), Cumbungi distribution in some reaches had expanded significantly. Following flooding in 2010–11, Cumbungi distribution was severely reduced, with it only being recorded in Wallpolla Creek during surveys the following year. Surveys over subsequent years documented the reappearance and slow increase of Cumbungi in most other reaches. During late 2016, extensive flooding occurred and this survey (November 2017) represents the first survey post flooding.

7.2 Ecological objectives

There are no ecological objectives relating directly to Cumbungi distribution across waterways of LMW. As such, an objective has been adopted that aims to;

limit Cumbungi dominance.

The basis for this objective stems from the overarching Environmental Water Management Plan (MDBA 2012b; MDBC 2007) ecological objective ‘Increase diversity and abundance of wetland aquatic vegetation’. It is thought that by limiting the abundance of cumbungi, other wetland aquatic vegetation can increase in diversity and abundance. However, it must be noted that changes in aquatic vegetation diversity and abundance are not monitored.

7.3 Methods

Initial channel surveys covering a total of 78 km (across 10 reaches), were made of representative sections of the primary anabranch channels and the adjacent River Murray in spring 2006. Coverage was extended to a total of 87.6 km across 11 reaches during the following monitoring period (summer 2007). Since 2007, monitoring has been undertaken in summer 2008, spring 2009, 2010, 2011, 2012, late winter 2013, spring 2015 and most recently spring 2017. Monitoring was not undertaken in 2014 and 2016 due to discontinuation of The Living Murray Program and omission of the Cumbungi monitoring component respectively. Additionally, the Mularoo and Toupnein Creeks were not assessed in 2010 due to restricted access.
Each bank of the watercourse was traversed and each Cumbungi stand was measured and the location recorded. A single stand was defined as being >1 m away from any other *Typha spp.* plant and Cumbungi was only measured if it was within 1 m of the waterline at normal weir operation level. Photo point/s, set up to capture Cumbungi in each reach, are re-taken annually during surveys.

Comprehensive detail on Cumbungi condition monitoring method is available in the Condition Monitoring Program design for the LMW icon site (Huntley et al. 2016a).

### 7.3.1 Analysis

For each reach, lengths of individual stands was summed to determine the total length of cumbungi. By dividing the total length of Cumbungi by two times the length of the reach (to account for both sides of the bank), the percent of bank covered by Cumbungi can be determined. Percent bank cover was compared between years to determine if condition (as measured by the variable/indicator ‘distribution’) is increasing, decreasing or stable.

### 7.3.2 Indices and points of reference

At this time, no points of reference have been finalised for Cumbungi distribution at LMW. Robinson (Robinson 2014b) made some initial comments toward this; however, further thought and investigation has not been made toward a final objective. At this time, Cumbungi distribution is reported against the ecological objective (see Section 7.2).

### 7.4 Results

Survey results from 2017 indicate that Cumbungi distribution has been substantially reduced at all reaches since it was last assessed in 2015 (Table 7.1). During 2017, Cumbungi was detected in four of the 11 reaches; Wallpolla Creek, Upper Lindsay River, Lock 7 and Lock 9. Two individual plants were recorded from Lock 9 (}
Table 7.2), a total of 2.7 m in Upper Lindsay River and 19.4 m in Lock 7. Wallpolla Creek retained the highest cover of Cumbungi (0.74%) in 2017, continuing an upward trend apparent since 2011 following extensive natural flooding. While the percent cover of Cumbungi was down compared to the last survey in Wallpolla Creek (2015), there was an increase in the number of stands. This was due to the re-shooting of a number of new plants that were distinct from each other, within the confines of old stands (having dead above-ground biomass visible).
Table 7.1 Cumbungi distribution across all reaches and survey years as percent bank cover, including reach lengths surveyed (N/A indicates not surveyed).

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<tr>
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### Table 7.2 Total number of Cumbungi stands by reach and year (N/A indicates not surveyed).

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7.5 Discussion

Flooding during late 2016 resulted in a decline in Cumbungi abundance across LMW. Elevated floodwater acts by ‘drowning’ Cumbungi plants, disrupting access to air and sunlight, causing aboveground biomass to die. Flooding also has a secondary effect, where elevated water velocity can flatten plants, driving them deeper underwater and also dislodging plants from the substrate though scouring. Flooding during summer 2010–11 caused a similar result, with Cumbungi distribution significantly reduced across the icon site following the highest recorded distribution in almost all reaches during the 2010 survey.

Following 2010–11 flooding, Wallpolla Creek was the only site to maintain some Cumbungi to the following survey. This was also apparent during the current survey (following flooding in 2016), with Cumbungi still present in Wallpolla Creek. It is thought that the persistence of Cumbungi in this location is an artefact of local hydrology. Wallpolla Creek is located immediately upstream of Lock 9; as such, when it floods, water depth does not increase to the extent as if it was located on the downstream side of a weir (1.5 m above weir pool level on upstream side compared with 3.9 m above weir pool level on downstream side; (MDBA 2017)). Consequently, water is not as deep for as long in Wallpolla Creek which may allow some of the taller Cumbungi plants to survive flooding. The same explanation may be used for the persistence of Cumbungi in Lock 7, with these stands located less than 1 km upstream of Lock 7.

In most instances, stands recorded in 2017 were growing in locations where old stands were present, as evident by dead Cumbungi stalks. This suggests that most plants have regenerated from belowground energy stores, or rhizomes. The few stands that were growing in locations with no previous Cumbungi evident generally comprised of one or two individual plants less than 50 cm tall. This suggests germination following flooding the previous year.

Over the next year, without further flooding, Cumbungi extent would be expected to increase naturally. Specifically, in Wallpolla Creek the number of total stands may decline as many smaller stands join, creating single larger stands. Expansion of Cumbungi at such low abundance should be considered positive as it increases diversity (some reaches currently have no Cumbungi) and results in increased habitat for various species. It only becomes an issue when it forms monospecific stands that span waterways. As such, when developing points of reference, ‘no cumbungi’ is not a realistic target as it makes up a significant part of the native riparian vegetation.

The established photo points for Cumbungi at all reaches across LMW no longer represent Cumbungi distribution and growth. The photo points were initially established based on Cumbungi stands at that time. Since flooding in 2010–11 and again in 2016 it is evident that Cumbungi is no longer present at any of the photo point locations. As such, it is recommended that additional photo points are established that incorporate live Cumbungi stands. This should be re-assessed annually, so when a stand is no longer apparent within a photo point, an additional photo point, incorporating a live stand is established.

As no point of reference currently exists for cumbungi, distribution is reported against the adopted ecological objective of ‘limit Cumbungi dominance’. Due to the reduction of Cumbungi from the previous survey, this objective is currently being met at LMW.
8 Fish

AUTHOR: DAVID WOOD, LOUISE ROMANIN & PAUL BROWN

8.1 Introduction

The installation of weirs across the lower Murray River has created a series of slow-flowing, deep weir pools with relatively constant water level and uniform hydraulic conditions in the main river channel. The hydrology of the Lindsay-Mulcra-Wallpolla (LMW) anabranch systems is regulated by the weirs of Locks 6, 7, 8, 9 and 10 and a miscellany of earthen and concrete structures at key anabranch effluent points. This results in a significant change from the natural flow regime and it is generally believed that native fish populations have undergone serious decline as a result (MDBC 2002). The Living Murray Program endeavours to benefit native fish through the modification of infrastructure and manipulation of flows to restore habitat quality and fish passage. Engineering works in the Lindsay River, Mullaroo Creek, Lake Wallawalla, Webster’s Lagoon and Potterwalkagee Creek have been completed to provide water management options for the benefit of native fish.

Condition monitoring reports on the change in environmental condition at the icon site scale including changes resulting from the implementation of works programs and the application of environmental water.

8.2 Ecological objectives

The Living Murray Environmental Water Management Plan (MDBA 2012b) ecological objective for fish at LMW is to:

*Increase abundance, diversity and extent of distribution of native fish.*

As a first step in The Living Murray (TLM) Program’s refinement process, which aims to better report on the ecological condition of icon sites in line with recommendations of the program review (Robinson 2013), an adopted objective was suggested,

*Maintain native fish populations, their relative abundance and diversity*

8.3 Methods


A nested sampling design, consisting of sites within reaches within macrohabitats, was used to assess the condition of fish assemblages across LMW Islands. At the highest level of sampling, macrohabitats were defined as riverine (i.e. the Murray River), larger no/slow-flow anabranch, faster-flowing (connecting) channel (between the Murray River and the anabranches) and wetland (Table 8.1). Four riverine, five anabranch, two channel reaches and one wetland were sampled. Within each reach, three sites were sampled, with the exception of Dedman’s Creek, for which only two sites were available due to its short length. Comprehensive details of the methods used and sites sampled are contained in Huntley et al (2016a).
Table 8.1. Location of fish sampling sites within each macrohabitat and reach.

<table>
<thead>
<tr>
<th>Macrohabitat</th>
<th>Location</th>
<th>Reach</th>
<th>Site codes</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>Lock 7 weir pool (L7)</td>
<td>L7.1, L7.2, L7.3</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Lock 8 weir pool (L8)</td>
<td>L8.1, L8.2, L8.3</td>
</tr>
<tr>
<td>Anabranch (no/slow flow)</td>
<td>Lindsay</td>
<td>Lower Lindsay River (LLR)</td>
<td>LLR.1, LLR.2, LLR.3</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Upper Lindsay River (ULR)</td>
<td>ULR.1, ULR.2, ULR.3</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Toupnein Creek (TC)</td>
<td>TC.1, TC.2, TC.3</td>
</tr>
<tr>
<td></td>
<td>Mulcra</td>
<td>Potterwalkagee Creek (PC)</td>
<td>PC.1, PC.2, PC.3</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Lower Wallpolla Creek (LWC)</td>
<td>WMC.1, WMC.2, WMC.3</td>
</tr>
<tr>
<td>Channel (fast flow anabranch)</td>
<td>Wallpolla</td>
<td>Dedman’s Creek (DC)</td>
<td>DC.1, DC.2, –</td>
</tr>
<tr>
<td></td>
<td>Lindsay</td>
<td>Mullaroo Creek (MC)</td>
<td>MC.1, MC.2, MC.3</td>
</tr>
<tr>
<td>Wetland</td>
<td></td>
<td>Webster’s Lagoon (WL)</td>
<td>WL.1, WL.2, WL.3</td>
</tr>
</tbody>
</table>

8.4 Indices and points of reference

Following detailed investigation of the suitability and sensitivity, a suite of indices were recommended as pertinent to the LMW icon site fish community (Brown et al. 2016; Huntley et al. 2016a).

Expectedness Index - For each reach, the proportion of sites where species richness exceeds the reference-level for ‘expected richness’. This effectively combines indices for diversity and expectedness. Reference values follow RC-F scores determined in Huntley et al (2016b) and are presented below.

\[ P_{\text{Expected}} = 6.7 \text{ (all macrohabitats)} \]

Nativeness Index - For each site, the proportion of fish biomass that is comprised of native species.

\[ P_{\text{Native}} = 0.7 \]

Recruitment Index - For each site, the count of native fish recruits as a proportion of the total count for all the native species.

\[ P_{\text{Recruits}} = 0.5 \]

In this publication we report against these indices and their reference values as recommended in the most recent review of TLM methods (Brown et al. 2016).

8.4.1 Index calculation

Below we present the workings involved to calculate each of the indices.

P Expected

Reference level is the expected icon-site species richness based on the combined data set from 2007–08 to 2016–17 (Brown et al. 2016). Values of P Expected can only lie between zero and one. A linear mixed model was fitted to the data using the lme4 package in R (Bates et al. 2014), such that “Year” is a fixed effect; and “Reach” nested within “Habitat,” are random effects (i.e., representative
of a broader distribution of these elements in the population of sites that is the icon site) (Brown et al. 2016). Data was analysed from reaches categorised as either ‘riverine’, ‘slow-flow anabranch’, or ‘fast-flow channel’. Data from Webster’s Lagoon was excluded as this was the only wetland sampled and occurred only periodically (when inundated). Annual mean estimates (±95% confidence interval) were plotted for each survey-year 2007–08 to the most recent, 2016–17. Post hoc comparisons of means for year-to-year change (Tukey Contrasts) were estimated using the multcomp package in R (Hothorn et al. 2008).

**P Native**

Most fish were measured and weighed at capture. Some fish (e.g. small-bodied species), were unable to be accurately weighed in the field. For those fish that were measured but not weighed, we estimated the weight ($\bar{W}$, grams) based on:

$$\bar{W} = 10[a + (b \cdot \log(L/c))]$$

Where $a$ and $b$, respectively, represent the constant and slope of the exponential weight-for-length curve ($W=al^b$) 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, 50 individuals were measured and weighed, as they were encountered, across all the replicates. The remainder in any replicate, were simply counted.

Because unmeasured and unweighed individual fish were present in some replicates in previous survey-years, we estimated the total biomass ($B'$) from the product of the species’ mean estimated weights ($\bar{W}$) and the total counts ($n$) of all individuals. (i.e. $B' = \bar{W}n$). The biomass of native species as a proportion of the total biomass (P Native) was calculated for each replicate. Where a species was present in a replicate with no measured representatives in that replicate, mean species’ weight for that sampling year was used. If no mean recorded weight was available for that species for that year (i.e. observations only), mean weight from all years was used.

Although calculated as a proportion at each site, the data is a continuous ratio of native biomass and total biomass (native + alien fish biomass), and must be analysed as such. Values can only lie between zero and one. A linear mixed model was fitted to the data using the lme4 package in R (Bates et al. 2014), such that “Year” is a fixed effect; and “Site” nested within “Reach,” nested within “Habitat,” are random effects (i.e. representative of a broader distribution of these elements in the population of sites that is the icon site) (Brown et al. 2016). Annual mean estimates (±95% confidence interval) were plotted for each survey-year 2007–08 to the most recent, 2016–17. Post hoc comparisons of means for year-to-year change (Tukey Contrasts) were estimated after adjustment for multiple-comparisons using the multcomp package in R (Hothorn et al. 2008).

**P Recruits**

Exotic species were excluded from calculations. A ‘recruit’ is defined as an individual with a total or fork length less than or equal to the size originally defining a young-of-year (i.e. age 0+ years) in the Sustainable Rivers Audit method (Robinson 2012) and reproduced in the recent TLM review document (Brown et al. 2016). Data here are counts as proportions. The true proportional nature of the data is taken advantage of by fitting a general linear mixed model, with a binomial error-distribution and a logit-link function, using the lme4 package in R (Bates et al. 2014). The model essentially performs a weighted ‘regression’ using the size of the denominator (count of native fish at a site) to weight the analysis by sample-size. Proportions from large sample sizes are more important to the model than proportions from small sample sizes. The model selected was such that “Year” is a fixed effect; and “Site” nested within “Reach,” are random effects (i.e. representative of a broader distribution of these elements in the population of sites that is the icon site) (Brown et al. 2016). Annual mean estimates (±95% confidence interval) were plotted for each survey-year 2007–
08 to the most recent, 2017–18. Post hoc comparisons of means for year-to-year change (Tukey Contrasts) were estimated using the multcomp package in R (Hothorn et al. 2008).

8.5 Results

During the present 2017–18 survey year, 31,722 fish were sampled. Ten species of native fish and five species of non-native fish were sampled using standard survey procedures (Scholz et al. 2007b). Samples were dominated by large numbers of native Carp gudgeon (*Hypseleotris* spp.), with Bony herring (*Nematalosa erebi*) being the next most common species (Table 8.2). With the exception of Carp gudgeon, being the highest abundance from all surveys for this species, all other species recorded abundancies that fell within previous recorded values.

Of particular note, for the iconic large-bodied natives, Golden perch (*Macquaria ambigua*) were recorded in their second highest numbers from all surveys (Table 8.2). While this is encouraging, the population distribution indicates a high proportion of adult fish with few juveniles (Figure 8.1). Murray cod (*Maccullochella peeli*) was also recorded in their second highest numbers across all surveys (Table 8.2 & Figure 8.3). Murray cod ranged in size from 99 mm to 1054 mm total length (TL) with majority of fish less than 200mm TL (Figure 8.2) suggesting recent recruitment.

![Figure 8.1](image1.png)

*Figure 8.1* Length frequency from a measured subsample of Golden perch (*n* = 98) from the 2017–18 monitoring year.

![Figure 8.2](image2.png)

*Figure 8.2* Length frequency from a measured subsample of Murray cod (*n* = 39) from the 2017–18 monitoring year.
### Table 8.2 Summary counts of all fish sampled at LMW Islands over ten sampling years as part of TLM Condition Monitoring (NB. 2014–15 not sampled).

<table>
<thead>
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</thead>
<tbody>
<tr>
<td><strong>Native</strong></td>
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<tr>
<td>Large-bodied</td>
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<tr>
<td>Bony herring</td>
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<td>26</td>
<td>6615</td>
<td>1288</td>
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<td>3768</td>
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<td>11</td>
<td>4</td>
<td></td>
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<td></td>
</tr>
<tr>
<td>Golden perch</td>
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<td>47</td>
<td>106</td>
<td>197</td>
<td>184</td>
<td>97</td>
<td>129</td>
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<td>219</td>
</tr>
<tr>
<td>Murray cod</td>
<td>73</td>
<td>49</td>
<td>27</td>
<td>4</td>
<td>2</td>
<td>3</td>
<td>14</td>
<td>33</td>
<td>7</td>
<td>50</td>
</tr>
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<td>Silver perch</td>
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<td>12</td>
<td>6</td>
<td>10</td>
<td>1</td>
<td>5</td>
<td>6</td>
<td>13</td>
<td>9</td>
<td>6</td>
</tr>
<tr>
<td>Spangled perch</td>
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<td>3</td>
<td>4</td>
<td></td>
<td></td>
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<tr>
<td><strong>Small-bodied</strong></td>
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<td></td>
<td></td>
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</tr>
<tr>
<td>Australian smelt</td>
<td>307</td>
<td>359</td>
<td>421</td>
<td>59</td>
<td>279</td>
<td>381</td>
<td>592</td>
<td>2497</td>
<td>1200</td>
<td>1945</td>
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<tr>
<td>Carp gudgeon</td>
<td>5261</td>
<td>5987</td>
<td>3517</td>
<td>712</td>
<td>1064</td>
<td>3950</td>
<td>5958</td>
<td>11625</td>
<td>3681</td>
<td>16320</td>
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<tr>
<td>Flathead gudgeon</td>
<td>362</td>
<td>710</td>
<td>201</td>
<td>53</td>
<td>734</td>
<td>1282</td>
<td>650</td>
<td>3408</td>
<td>1420</td>
<td>420</td>
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<tr>
<td>Dwarf flathead gudgeon</td>
<td>2</td>
<td>18</td>
<td>21</td>
<td>6</td>
<td>8</td>
<td>12</td>
<td></td>
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<td>325</td>
<td>799</td>
<td>267</td>
<td>607</td>
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<td>Un-specked hardyhead</td>
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<td>2129</td>
<td>1365</td>
<td>460</td>
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<td>101</td>
<td>74</td>
<td>4379</td>
<td>783</td>
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<tr>
<td><strong>Non-native</strong></td>
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<tr>
<td>Large-bodied</td>
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<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Common carp</td>
<td>288</td>
<td>152</td>
<td>399</td>
<td>1552</td>
<td>1998</td>
<td>626</td>
<td>691</td>
<td>508</td>
<td>8355</td>
<td>753</td>
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<tr>
<td>European perch</td>
<td>24</td>
<td>8</td>
<td>31</td>
<td>6</td>
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<td>2</td>
<td>1</td>
<td>4</td>
<td>15</td>
<td></td>
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<tr>
<td>Goldfish</td>
<td>81</td>
<td>49</td>
<td>330</td>
<td>172</td>
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<td>13</td>
<td>152</td>
<td>337</td>
<td>850</td>
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<td>Oriental weatherloach</td>
<td>2</td>
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<td></td>
<td></td>
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<tr>
<td><strong>Small-bodied</strong></td>
<td>3393</td>
<td>1092</td>
<td>2616</td>
<td>23023</td>
<td>85</td>
<td>1264</td>
<td>1260</td>
<td>2943</td>
<td>1697</td>
<td>4505</td>
</tr>
<tr>
<td>Eastern mosquito</td>
<td>3393</td>
<td>1092</td>
<td>2616</td>
<td>23023</td>
<td>85</td>
<td>1264</td>
<td>1260</td>
<td>2943</td>
<td>1697</td>
<td>4505</td>
</tr>
<tr>
<td><strong>Grand Total</strong></td>
<td>12817</td>
<td>10723</td>
<td>15697</td>
<td>27605</td>
<td>7876</td>
<td>11534</td>
<td>18414</td>
<td>34888</td>
<td>31601</td>
<td>31722</td>
</tr>
</tbody>
</table>
Figure 8.3 Relative abundance (total sample size) and average weight (g) of Murray cod sampled during monitoring years 2007–08 to 2017–18.

8.5.1 Species Richness

The proportion of sites exceeding reference-level for expected richness of 6.7 per site in the current monitoring year (2017–18) is 0.66 (Figure 8.4). This continues the decline occurring from the highest proportion in 2015–16 but is still the third highest proportion from all monitoring years. The difference between 2016–17 and 2017–18 was not deemed significantly different (P>0.05). Large confidence intervals indicates high variability across the icon site for each monitoring year.

Figure 8.4 The mean proportion of sites in each survey year (±95% Confidence intervals) where the species richness exceeds the reference level of expected species-richness (based upon potential richness combining all data to 2016–17). Statistically significant changes between successive years surveyed are marked as ** P<0.01 or * P<0.05.

8.5.2 Proportion of native fish biomass — P Native

The current monitoring year saw a highly significant increase (P<0.001) in the proportion of native fish biomass from the previous year (2016–17) (Figure 8.5). Even though an increase occurred, the reference level (0.7) was not achieved for 2017–18. The reference level has only been achieved on a single occasion (2015–16) across all monitoring years.
Figure 8.5 The estimated mean proportion (±95% confidence intervals) of fish biomass that is native fish biomass at a site (P Native) during fish surveys of LMW icon site between 2007–08 and 2017–18. Statistically significant changes between successive years surveyed are marked as *** (P<0.001).

8.5.3 Proportion of native fish recruitment — P recruits

There was an increase in the proportion of native fish recruits between the previous monitoring year (2016–17) and this monitoring year (2017–18) (Figure 8.6). There was a highly significant difference (P<0.001) between these years, as has been the case through most pairs of years. This indicates large annual variability. Even though the proportion of native fish recruits increased, the reference level (0.5) was not achieved for 2017–18. Recruitment of native fish between 2011–12 and 2015–16 (inter-flood period) was highest of all monitoring years and represents the only time the reference value has been achieved though TLM condition monitoring at LMW.

Figure 8.6 The estimated proportion (±95% confidence intervals) of native fish that are ‘recruits’ (P recruit) each survey year during fish surveys of LMW icon site between 2007–08 and 2017–18. Statistically significant changes between successive years surveyed are marked as *** (P<0.001).
8.7 Discussion

Surveys across LMW during 2017–18 sampled a relatively high number of fish compared with previous surveys. There were no new species sampled; however, Oriental weatherloach (Misgurnus anguillicaudatus) were sampled after being absent from sampling since the 2010–11 monitoring year. This also corresponded with a significant flood that saw the rapid expansion in the range of this species (Fredberg et al. 2014). Species absent from sampling during 2017–18 include Spangled perch (Leiopotherapon unicolor) and Dwarf flathead gudgeon (Philypnodon macrostomus). The absence of either of these species is not unexpected as Spangled perch are rare in the southern basin and usually only appear following flooding that ‘washes’ them down (Ellis et al. 2015; Lintermans 2007). Dwarf flathead gudgeon are scarce through the majority of the Murray-Darling Basin (Lintermans 2007) and have not been recorded in large numbers across LMW previously.

Following a low number of Murray cod (n=7) sampled the previous monitoring year (2016–17), this year (2017–18) recorded the second highest number across all monitoring years (n=50). Of particular interest is that the majority of these fish are new recruits. Such an occurrence has never been recorded previously at LMW during condition monitoring surveys and is an important first step in rebuilding the population following increased mortality due to blackwater events during the previous two floods. Larger Murray cod (>500 mm Total Length) were scarce; however, their presence indicate that many survived hypoxic conditions during the 2016–17 flood and are beginning to repopulate this region.

Following the highest number of Golden perch recorded during 2016–17, the number sampled during this monitoring year (2017–18) was still high, being the second highest number sampled over the entire monitoring program. Much like last year, nearly all of the Golden perch this year were adult (>200 mm Total Length). Only a single juvenile was sampled during 2017–18. It is highly likely (based on location and timing) that this individual was released as part of an angler stocking program in that river reach. The lack of Golden perch recruitment across LMW is of some concern; however, studies suggest that breeding of this species is episodic and strongly driven by flow pulses (Sharpe 2011; Zampatti & Leigh 2013).

Common carp (Cyprinus carpio) and to a lesser extent Goldfish (Carassius auratus), were not as prevalent during 2017–18 as they were following flooding during 2016–17. The number of Eastern mosquitofish (Gambusia holbrooki) were high but not to the extent seen during the 2010–11 flood. While perhaps not as abundant as previously recorded, non-native species still comprise a significant proportion of the population as biomass. This is highlighted by the P native index, with native fish, while far more numerically dominant, only making up around 60% of the total biomass.

With regards to the objective; Maintain native fish populations, their relative abundance and diversity, we can say that this objective has partially been met. Support for the objective comes from the fact that the proportion of native fish recruits and the mean proportion of fish biomass have both increased from the previous monitoring year. Also, in terms of native species abundance, most species are well represented and all fall within the range of individuals previously recorded (except Carp gudgeon, recorded in their highest number in 2017–18). Species richness objectives; however, are not being met. The mean proportion of sites exceeding the reference level of expected species richness was lower than the previous monitoring year. This indicates a reduction in overall diversity and distribution across the icon site. Additionally, neither the proportion of native fish recruits nor the mean proportion of fish biomass reached their respective reference levels for 2017–18.

The objective for fish at LMW icon site, Maintain native fish populations, their relative abundance and diversity was only partially met in 2017–18. While a number of indices (P native & P recruit) improved from the previous year, reference levels were not achieved. In addition, the indices relating to P expectedness declined from the previous year.
9 Birds

AUTHOR: RICHARD LOYN AND GARRY CHEERS

9.1 Introduction

The icon sites include ephemeral wetlands that provide valuable habitat for waterbirds when they are flooded. Environmental flows are expected to enhance those habitat values, and waterbirds are included in programs for monitoring the condition of the icon sites. In the Victorian Mallee, waterbirds have been monitored regularly at seven wetlands at Lindsay-Mulcra-Wallpolla (LMW) icon site (Henderson et al. 2014). This report presents results from this monitoring program for the 2017-18 year (spring 2017 and autumn 2018).

9.2 Ecological objective

Ecological objectives have been set for each icon site, as summarised by (Henderson et al. 2014). Overarching objectives are to;

- Provide habitat for a range of waterbird species, including migratory species and colonial nesters.

More specific objectives for LMW are to;

- Provide occasional breeding and roosting habitat for colonial waterbirds

- Provide habitat suitable for migratory birds, especially species listed under the JAMBA, CAMBA and RoKAMBA agreements (between Australia and Japan, China and the Republic of Korea respectively).

The objective for breeding waterbirds required separate targeted monitoring programs (e.g. (Willis et al. 2015)), as breeding colonies are highly localised and not always in predictable sites. This report focuses on the monitoring program as it relates to the second objective and the overarching objective about waterbirds.

9.3 Sampling method

Waterbirds were counted at each site using a timed 20-minute search from a specified vantage point in accordance with The Living Murray Standard Waterbird Assessment Approach (http://www.birdlife.org.au/documents/ATL-Starter-Kit-2012.pdf). This involves recording numbers of all bird species observed from the vantage point in the set period, following the area-search method devised for bush-birds by Loyn (1986). Small increases in time spent were allowed when large numbers of birds were present, up to 30 minutes total, and 30 minutes has become the standard for these waterbird counts over recent years (Loyn et al. 2017). The method has been used for monitoring these sites since 2005 (Cook & Jolly 2010, 2011; Henderson et al. 2014; Henderson et al. 2013; Henderson et al. 2012; Lown et al. 2017; Willis et al. 2015).

The present report considers waterbird counts made from spring 2017 to autumn 2018. Most wetlands were surveyed once in spring (September-October 2017) and again in autumn (March 2018). Lake Wallawalla was also surveyed in January 2018 as part of a separate program of intervention monitoring for that wetland. Surveys were undertaken on three dates at Lake Wallawalla. All other sites were surveyed twice: once in September-October 2017 and once in March 2018.
9.4 Indices and points of reference

Suitable indices and points of reference have yet to be refined for wetland birds at the LMW icon site.

9.5 Results

All results are summarised in Table 9.1 for Lake Wallawalla and elsewhere on LMW in Table 9.2. Results of individual waterbird counts in 2017-18 are presented in Part B; Table 9.1 (Lake Wallawalla) and Table 9.2 (elsewhere on LMW). The guild classification is shown in Part B, Table 9.3, following a previous report (Loyn et al. 2017).

Lake Wallawalla received strong environmental flows in spring 2017 and water levels receded gradually over subsequent months (Table 9.2). Elsewhere in the LMW system, Lilyponds was 80-90% full in both seasons, Mulcra Horseshoe was 70% full in spring but dry in autumn, Stockyards was dry on both visits, and Wallpolla and Webster’s Lagoons were dry in spring and partially filled in autumn (the downstream western half in both cases).

**Table 9.1** Total numbers of waterbird species observed on two standard searches at Lake Wallawalla, spring 2017 and autumn 2018.

<table>
<thead>
<tr>
<th>Waterbird species</th>
<th>Spring 2017</th>
<th>Autumn 2018</th>
<th>Waterbird species</th>
<th>Spring 2017</th>
<th>Autumn 2018</th>
</tr>
</thead>
<tbody>
<tr>
<td>Freckled Duck</td>
<td>0</td>
<td>52</td>
<td>Straw-necked Ibis</td>
<td>3</td>
<td>7</td>
</tr>
<tr>
<td>Black Swan</td>
<td>6</td>
<td>249</td>
<td>Royal Spoonbill</td>
<td>0</td>
<td>10</td>
</tr>
<tr>
<td>Australian Shelduck</td>
<td>7</td>
<td>1</td>
<td>Yellow-billed Spoonbill</td>
<td>64</td>
<td>192</td>
</tr>
<tr>
<td>Australian Wood Duck</td>
<td>492</td>
<td>1592</td>
<td>Black-tailed Native-hen</td>
<td>0</td>
<td>380</td>
</tr>
<tr>
<td>Pink-eared Duck</td>
<td>0</td>
<td>203</td>
<td>Eurasian Coot</td>
<td>0</td>
<td>9</td>
</tr>
<tr>
<td>Grey Teal</td>
<td>78</td>
<td>810</td>
<td>Black-winged Stilt</td>
<td>0</td>
<td>25</td>
</tr>
<tr>
<td>Pacific Black Duck</td>
<td>0</td>
<td>10</td>
<td>Red-necked Avocet</td>
<td>156</td>
<td>0</td>
</tr>
<tr>
<td>Australian Pelican</td>
<td>25</td>
<td>170</td>
<td>Red-capped Plover</td>
<td>32</td>
<td>12</td>
</tr>
<tr>
<td>White-necked Heron</td>
<td>0</td>
<td>9</td>
<td>Red-kneed Dotterel</td>
<td>0</td>
<td>8</td>
</tr>
<tr>
<td>Eastern Great Egret</td>
<td>0</td>
<td>8</td>
<td>Masked Lapwing</td>
<td>42</td>
<td>49</td>
</tr>
<tr>
<td>White-faced Heron</td>
<td>6</td>
<td>21</td>
<td>Caspian Tern</td>
<td>24</td>
<td>1</td>
</tr>
<tr>
<td>Australian White Ibis</td>
<td>0</td>
<td>24</td>
<td>Silver Gull</td>
<td>0</td>
<td>1</td>
</tr>
</tbody>
</table>
Table 9.2 Total numbers of waterbird guilds, wetland area inundated and waterbird density observed at Lake Wallawalla and all other LWM wetlands, spring 2017 and autumn 2018.

<table>
<thead>
<tr>
<th>Waterbird guilds</th>
<th>Lake Wallawalla</th>
<th>All other LMW sites</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Spring 2017</td>
<td>Autumn 2018</td>
</tr>
<tr>
<td>Coots</td>
<td>0</td>
<td>9</td>
</tr>
<tr>
<td>Dabbling ducks</td>
<td>78</td>
<td>820</td>
</tr>
<tr>
<td>Diving ducks</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Filter-feeding ducks</td>
<td>0</td>
<td>255</td>
</tr>
<tr>
<td>Fish-eaters (cormorants, darter &amp; pelican)</td>
<td>25</td>
<td>170</td>
</tr>
<tr>
<td>Grazing ducks</td>
<td>492</td>
<td>1593</td>
</tr>
<tr>
<td>Grebes</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Gulls</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>Large wading birds</td>
<td>73</td>
<td>271</td>
</tr>
<tr>
<td>Shorebirds</td>
<td>230</td>
<td>94</td>
</tr>
<tr>
<td>Swans</td>
<td>6</td>
<td>249</td>
</tr>
<tr>
<td>Terns</td>
<td>24</td>
<td>1</td>
</tr>
<tr>
<td>Waterhens</td>
<td>0</td>
<td>380</td>
</tr>
<tr>
<td>All grazers (coot, hens, swan, grazing ducks)</td>
<td>498</td>
<td>2231</td>
</tr>
</tbody>
</table>

**All waterbirds**

<table>
<thead>
<tr>
<th></th>
<th>Lake Wallawalla</th>
<th>All other LMW sites</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Spring 2017</td>
<td>Autumn 2018</td>
</tr>
<tr>
<td></td>
<td>928</td>
<td>3843</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Waterbird species</th>
<th>Spring 2017</th>
<th>Autumn 2018</th>
<th>Waterbird species</th>
<th>Spring 2017</th>
<th>Autumn 2018</th>
</tr>
</thead>
<tbody>
<tr>
<td>Australian Shelduck</td>
<td>4</td>
<td>4</td>
<td>Pied Cormorant</td>
<td>3</td>
<td>3</td>
</tr>
<tr>
<td>Australian Wood Duck</td>
<td>39</td>
<td>39</td>
<td>White-necked Heron</td>
<td>1</td>
<td>4</td>
</tr>
<tr>
<td>Pink-eared Duck</td>
<td>0</td>
<td>10</td>
<td>Eastern Great Egret</td>
<td>0</td>
<td>6</td>
</tr>
<tr>
<td>Grey Teal</td>
<td>45</td>
<td>51</td>
<td>White-faced Heron</td>
<td>2</td>
<td>3</td>
</tr>
<tr>
<td>Pacific Black Duck</td>
<td>0</td>
<td>26</td>
<td>Nankeen Night Heron</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Blue-billed Duck</td>
<td>0</td>
<td>1</td>
<td>Australian White Ibis</td>
<td>5</td>
<td>3</td>
</tr>
<tr>
<td>Australasian Grebe</td>
<td>3</td>
<td>72</td>
<td>Yellow-billed Spoonbill</td>
<td>1</td>
<td>16</td>
</tr>
<tr>
<td>Hoary-headed Grebe</td>
<td>5</td>
<td>0</td>
<td>Black-tailed Native-hen</td>
<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Great Crested Grebe</td>
<td>0</td>
<td>6</td>
<td>Eurasian Coot</td>
<td>0</td>
<td>11</td>
</tr>
<tr>
<td>Australasian Darter</td>
<td>1</td>
<td>1</td>
<td>Black-fronted Dotterel</td>
<td>4</td>
<td>2</td>
</tr>
<tr>
<td>Little Pied Cormorant</td>
<td>0</td>
<td>12</td>
<td>Red-kneed Dotterel</td>
<td>5</td>
<td>0</td>
</tr>
</tbody>
</table>

Table 9.3 Total numbers of waterbird species observed on standard sites at LMW, spring 2017 and autumn 2018.
9.5.1 **Numbers of waterbirds in selected wetlands**

Total numbers of waterbirds counted on the selected wetlands were substantially higher in autumn than in spring (Table 9.2). The highest counts were at Lake Wallawalla, where total waterbird numbers rose from 928 in spring 2017 to 1846 in summer and 3843 in autumn 2018. This was partly due to the large size of that lake (880 ha) compared with other wetlands.

Waterbird densities (total waterbirds counted per ha of surface water) ranged from zero in most dry wetlands to 5.5 at Lake Wallawalla and 11.1 at Lilyponds in autumn 2018 (the latter being a small wetland, 90% full at the time when many other small wetlands nearby had dried). Even higher density values were recorded at two small wetlands (Stockyard and Wallpolla) in autumn and spring respectively when they were virtually dry, with just two pools of water in the latter case. However, these values are not considered reliable estimates as bird numbers can be expected to fluctuate rapidly at such small refuge sites, and our estimates of water cover are not precise.

9.5.2 **Waterbird breeding**

No signs of colonial breeding events were observed in the current year (though the current program was not designed to seek such evidence). Evidence of breeding was noticed for two species of grebe at Wallpolla Lagoon, where broods of five juvenile Australasian Grebes and three juvenile Great Crested Grebes were recorded. It is likely that many more common waterbirds (e.g. Grey Teal and Australian Wood Duck) bred inconspicuously at or near many of these wetlands during the year.

9.5.3 **Guild and species composition**

Grazing ducks were the most numerous guild, constituting 42% of waterbirds observed across all standard sites in both seasons. They were represented mainly by Australian Wood Duck, which graze extensively from swards of green terrestrial vegetation on wetland margins and were found mainly on grassy flats around Lake Wallawalla, with far fewer elsewhere. A few pairs of Australian Shelduck were seen mainly along the Murray River, and were probably breeding in low numbers in nearby wetlands. The grazing duck guild was well represented at Lake Wallawalla but otherwise widely though sparsely distributed.

Dabbling ducks were the second most numerous waterbird guild, constituting 20% of the waterbird community. Birds in this guild take animal and vegetable food in shallow water by upending or dabbling. Grey Teal were by far the most numerous species in this guild, with lower numbers of Pacific Black Duck. Grey Teal were found in varying numbers on most wetlands, and the largest concentrations were at Lake Wallawalla.

Large wading birds (e.g. herons, egrets, ibis and spoonbills) were widespread and common on many wetlands, constituting 7.5% of the waterbird community. Shorebirds were more local, with some species scattered in low numbers (Masked Lapwing and Black-fronted Dotterel) and others recorded locally in substantial numbers (Red-necked Avocet and Black-winged Stilt feeding from open water; Red-kneed Dotterels mainly on vegetated margins where water was advancing or receding). No transcontinental migratory shorebirds were observed this year.

Filter-feeding ducks were scarce (5%), but up to 203 Pink-eared Duck and 52 Freckled Duck were found at Lake Wallawalla in autumn, with some Freckled Duck seen earlier in summer (A. Greenfield pers. comm.). Just a few Pink-eared Duck were found elsewhere, with eight at Lilyponds and two at Wallpolla Lagoon.

Diving ducks were extremely scarce with only a single Blue-billed Duck recorded at Wallpolla Lagoon in autumn.

Black Swans (5%) were recorded only at Lake Wallawalla, where there were six in spring and 249 in autumn. Eurasian Coot (0.4%) were recorded only in autumn, when remarkably small numbers were
found at Lilyponds (7), Wallpolla Lagoon (4) and Lake Wallawalla (9). Both of these species feed on aquatic vegetation; swans by reaching below the surface (upending if necessary) and coot by diving. **Waterhens** (7.5%) were represented at these sites by one species, the Black-tailed Native-hen, which shelters in dense vegetation but may feed in more open situations along wetland margins, walking or running rather than swimming to find food. Two Black-tailed Native-hens were seen at Lilyponds in autumn, when an extraordinary concentration of 380 Black-tailed Native-hens was found on vegetated margins of Lake Wallawalla. Together with the grazing ducks (discussed above), these essentially vegetarian guilds constituted 55% of waterbirds observed during these two assessments.

Our guild of waterhens includes three relatives of the Eurasian Coot (Purple Swamphen, Dusky Moorhen and Black-tailed Native-hen). They share the coot’s mainly vegetarian diet but generally feed from the surface of shallow water (Dusky Moorhen) or from vegetation growing on the landward margins of wetlands. Purple Swamphens and Dusky Moorhen were not recorded during the waterbird surveys, although there is a resident population of Purple Swamphen at Kings Billabong near Mildura. Flocks of Black-tailed Native-hens were observed at a number of wetlands as waters receded, especially at Lake Wallawalla where 184 were counted in January and 380 in March 2018 an extraordinary concentration. The receding margins of this lake clearly offer an important food source for birds that feed on green vegetation, including Australian Wood Duck (1592 in autumn) and Black-tailed Native-hen (380).

Fish-eating pelicans, cormorants and darters (4%) were much less numerous than at the Hattah Lakes. Up to 170 Australian Pelicans were counted at Lake Wallawalla (in autumn) but other species in the guild were remarkably scarce, with records of just six Little Pied Cormorants, three Pied Cormorants and an Australasian Darter. The two black cormorants (Great Cormorant and Little Black Cormorant) were not seen on these counts, although both can be common along the Murray River and nearby wetlands.

Grebes were even rarer, constituting 1.7% of the waterbird community. Australasian Grebes were found mainly on small wetlands, and Hoary-headed Grebes were only observed once (five at Lilyponds in spring).

Gulls and terns were also remarkably scarce, and on the standard wetlands they were not recorded except at Lake Wallawalla (with 24 Caspian Terns in spring and one in autumn, and one Silver Gull in autumn). On previous visits, Caspian Terns were sometimes seen fishing along the Murray River nearby.

**9.5.4 Lake Wallawalla**

Lake Wallawalla attracted a larger and more diverse waterbird community than any of the other wetlands examined in this study. Two herbivorous waterbird species proved to be particularly numerous in autumn 2018 with counts of 1592 Australian Wood Duck and 380 Black-tailed Native-hens feeding from the extensive green swards of vegetation on the surrounding flats as waters receded. The lake also attracted substantial numbers of Australian breeding shorebirds (notably up to 156 Red-necked Avocets) and filter-feeding ducks (notably up to 54 Freckled Duck, listed as Endangered in Victoria). The absence of Grebes from this lake, during this year’s survey, was remarkable. A rare land bird species (Grey Falcon), was seen flying across Lake Wallawalla during the September count; this species is listed as Vulnerable globally and Endangered in Victoria.

**9.5.5 Elsewhere in the Lindsay-Mulcra-Wallpolla system**

Most of the remaining wetlands supported similar waterbird communities to the Hattah Lakes (with the dabbling duck Grey Teal as the most common species), except for lower numbers of fish-eating pelicans, cormorants and darters. Several small wetlands attracted concentrations of Australian-breeding shorebirds as they dried. Notable records included nine Red-kneed Dotterels and 10 Black-fronted Dotterels at Webster’s Lagoon in autumn 2018.
9.6 Discussion

Dynamic patterns of fluctuation in waterbird numbers have been recognised for many years, in response to fluctuating availability of water over vast areas of the continent (Chambers & Loyn 2006; Frith 1982; Kingsford & Norman 2002; Marchant & Higgins 1993a; Marchant & Higgins 1993b). Hence the numbers of waterbirds at a given site is influenced by the availability of water elsewhere, as well as the current value of that particular site. Numbers of waterbirds can change rapidly as waters ebb and flow in different parts of the continent.

9.6.1 Patterns of response

Our results accord with the general pattern for previous years that waterbirds are attracted to new floodwaters wherever they occur, but the composition of guilds and species is influenced by local conditions (Loyn et al. 2017).

Lake Wallawalla continues to attract a larger number and greater diversity of waterbirds than the other wetlands studied. This is partly due to its large size (880 ha), but not wholly, as bird densities were higher than at many other wetlands. The topography of the lake encourages growth of low vegetation on the extensive flat plains as waters recede, attracting large numbers of herbivorous waterbirds, mainly of two species: Australian Wood Duck and Black-tailed Native-hen. The lake itself must support a diverse suite of invertebrate fauna, expected to be the main food source for most of the ducks that feed from the lake (Grey Teal, Pink-eared Duck, Freckled Duck, etc.), for the shorebirds that feed from its waters (Red-necked Avocet and Black-winged Stilt) or the surrounding shores (Masked Lapwing, Red-capped Plover and others), and for some of the long-legged wading birds such as Yellow-billed and Royal Spoonbills that feed from the shallows. The low numbers of fish-eating birds (and the extraordinary absence of grebes) must reflect a lower fish population than at the Hattah Lakes and this may have contributed positively to the abundance of other waterbird guilds. It was especially pleasing to find that the lake was able to attract a substantial number of Freckled Duck (52), as this is one of the iconic species that is often mentioned in relation to the Ramsar listing of both icon sites. Freckled Ducks are listed as Endangered in Victoria, and they represent an endemic Australian waterfowl genus, from an ancient lineage not found elsewhere in the world.

The other wetlands in the LMW group continue to attract waterbirds when they are flooded, mostly of the more common species. The scarcity of less common species reflects the small size of these wetlands and the consequent low diversity of habitats offered. Nevertheless, they make an important contribution to the waterbird populations of the region as a whole.

9.6.2 Implications for management

The results show that environmental flows can be useful to waterbirds wherever they are applied, as waterbirds made use of all wetlands fed by environmental flows and had previously been absent from those wetlands when they were dry (Loyn et al. 2017). However, environmental flows appear to have greater value in large wetlands such as Lake Wallawalla than in small wetlands such as the other lagoons in LMW. Flows into wetlands that have previously been dry (e.g. Lake Wallawalla) appear to have beneficial impact for more waterbird species than flows into wetlands that have retained water for longer periods and now support mature populations of fish. However, fish-eating birds may benefit more from the latter situation, as observed in the Hattah Lakes. Clearly, a mix of strategies is needed to provide habitat for the full suite of waterbird species in the broader landscape.

In defining management objectives for icon sites or Ramsar-listed wetlands, it has been customary to emphasise their potential for conserving threatened species or international migrants listed under international treaties. This may lead to perverse outcomes when those species form very small proportions of the waterbird community, and when the sites in question support very small.
proportions of the global species population. It has become clear that the sites considered in this report do not usually attract large numbers of threatened waterbird species and they rarely attract more than a handful of international migratory shorebirds (Loyn et al. 2017). From this perspective, it was pleasing that some threatened waterbird species were recorded in the current year, including a substantial flock of 52 Freckled Duck at Lake Wallawalla. The recent environmental flows have clearly delivered some benefits to those species.

We were surprised not to find at least a few migratory shorebirds at Lake Wallawalla and other wetlands and suspect that the rapid revegetation of mudflats as waters recede may limit the feeding opportunities for those species (which generally feed in flocks on open shorelines providing clear views of potential predators). Studies of the geological substrate and the invertebrate fauna of the mudflats would be needed to shed further light on the rarity of international migratory shorebirds at all these sites, compared with certain wetlands elsewhere in the Murray Valley that regularly attract hundreds of these birds (e.g. many of the lakes near Kerang and Swan Hill). It remains possible that several of the wetlands we examined could have the potential to support migratory shorebirds when suitable conditions generate the open wet muddy surfaces favoured by those birds.

The results show that these wetlands continue to provide habitat for a wide range of waterbird species. These happen to include some that are listed as threatened in Victoria and the record of up to 52 Freckled Duck at Lake Wallawalla is especially notable in this respect. Lake Wallawalla proved to be the outstanding habitat for waterbirds, attracting large numbers of a wide range of species.

In terms of more specific objectives, no colonial breeding was found in these wetlands on this occasion. Several species that breed colonially were observed, but not breeding (e.g. egrets). Some such species were found in substantial numbers, e.g. 170 Australian Pelicans at Lake Wallawalla. These birds were resting on the surrounding flats, showing that this wetland provides roosting habitat for these and other waterbirds.

No transcontinental migratory birds were observed on these assessments, although we do not rule out the possibility that some could use these wetlands in future years. Lake Wallawalla may have the best chance of providing habitat for such species in future.
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