The Living Murray Condition Monitoring at Lindsay, Mulcra and Wallpolla Islands 2015–16
Part A – Main Report

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Part A – Main Report

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

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Executive summary

This report details the condition monitoring undertaken at Lindsay, Mulcra and Wallpolla (LMW) Islands as part of The Living Murray (TLM) Condition Monitoring Program. Icon site condition monitoring has been developed to:

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

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

River Red Gum

As of 2015–16, it is deemed that the overall health of River Red Gum is being maintained above current targets, therefore the ecological objective *Maintain current condition and extent of River Red Gum communities to sustain species assemblages and processes typical of such woodland* at the LMW icon site is being met.

For 2015–16, the target of 85% of River Red Gum trees exceeding a crown extent score of ≥ 4 was met at LMW. This target has been consistently met since 2011–12.

Germination of River Red Gum seedlings was prolific at LMW during the one to two years prior to 2012–13. This is attributed to increased water availability and flooding in 2010–11. By 2015–16, many of these seedlings appeared to have died. However, seedlings (trees with < 15 cm diameter-at-breast-height) still comprise a significant proportion of the overall population.

Recruitment (to mature trees) has been greater than mortality (of mature trees) for River Red Gum at LMW since 2008–09.

Black Box

As of 2015–16, it is deemed that, although Black Box condition currently meets the ecological objective *Improve condition to sustain species assemblages and processes typical of Black Box woodland*, the overall health of the population is in decline.

The current condition of Black Box at LMW is of some concern. While the target of 80% of Black Box trees exceeding a crown extent score of ≥ 4 was met, Black Box swampy woodland and riverine chenopod woodland condition is in decline.

Black Box seedling numbers at LMW have been declining since 2009–10, with the majority of the population comprised of mature trees. The riverine chenopod woodland population is of some concern, as it has experienced a long trend of annual population loss. Recently (2013–14 and 2015–16), this has reversed, however, recovery of the population may take many consecutive years of net growth. Recovery is potentially threatened by the presence of a low number of seedlings at the pre-recruitment stage. The Black Box swampy woodland population is in better condition, with net population gain occurring annually since 2008–09.

Wetland vegetation communities

The ecological condition of wetland vegetation communities for the LMW icon site in 2015–16 met the overarching ecological objective for vegetation, *Increase the diversity, extent and abundance of wetland vegetation.*

The majority (eight of twelve) of wetlands surveyed at LMW in 2015–16 were long-dry. Three wetlands were intermittent-dry (had held water in the last two years) and one wetland, Lake Wallawalla, was drawing down after receiving environmental water in spring 2015. The latter four wetlands provided damp/wet habitats favourable for wetland vegetation.

A total of 109 wetland vegetation species were recorded at LMW in 2015–16, which was an increase from the previous monitoring year (93 species recorded in 2013–14). The most abundant species in 2015–16 was Common Sneezeweed (*Centipeda cunninghamii*), an amphibious herb that has emerged in response to recent inundation. Of the 109 species, 10 were not previously recorded at LMW over nine years of TLM wetland condition monitoring. The increase in species diversity and addition of species not previously recorded during this program are largely attributed to a response to recent inundation at four of the twelve wetlands surveyed.

Sixteen species recorded in 2015–16 were listed as having conservation significance in Victoria, and five of these were new to the TLM wetland dataset: *Atriplex vesicaria* subsp. *macrocystidia*, Inland Club-sedge (*Isolepis australiensis*), *Isolepis congrua*, Bottom Rush (*Lipocarpha microcephala*) and Green Copperburr (*Sclerolaena decurrens*). Environmental flows appear to be supporting flow-dependent rare plants. There is limited information about these species, largely because of their ephemeral nature. Targeted surveys (timed to coincide with the drawdown of wetlands following the delivery of environmental water) are recommended and would provide valuable information for the management of flow-dependent rare plants.

Floodplain vegetation communities

The condition of floodplain understorey communities at LMW in 2015–16 is that of a dry floodplain. All sites are considered long-dry (not inundated by overbank flows for more than two years) and were heavily dominated by drought-tolerant species in both River Red Gum and Black Box understorey communities. While the drying process is important for understorey vegetation communities in arid floodplains, intermittent inundation is required to sustain aquatic and amphibious plant species. River Red Gum and Black Box understorey communities would benefit from inundation in the near future. Proposed infrastructure at this icon site may benefit some of these communities.

A total of 59 floodplain plant species were recorded in 2015–16, 7 of which have conservation significance in Victoria. Two-spined Copperburr (*Sclerolaena uniflora*) was recorded for the first time in 2015–16 over nine years of TLM floodplain monitoring at LMW. This species is listed as rare in Victoria and was found on Wallpolla Island. The abundance of non-native species was consistently low in all monitoring years at LMW. In particular, non-native species were less than 5% of the proportion of abundance in Black Box understorey communities.

As part of an intervention monitoring program funded by the Mallee Catchment Management Authority, additional floodplain sites were established at Mulcra Island to specifically capture a response to environmental watering. Data were collected in 2011 (prior to environmental flows) and 2013 (following environmental flows), but have not yet been analysed. It is recommended that these data be analysed as they could provide valuable insight into the effect of environmental water on floodplain understorey vegetation condition.

Lignum

Surveys of Lignum across the icon site in 2015–16 showed no signs of significant recovery in the condition of Lignum communities.

Over the last decade of TLM condition monitoring, the LMW icon site has experienced severe drought (mid-1990s to 2009), followed by widespread flooding (late 2010–11). Areas of floodplain that were not inundated by overbank flooding received substantial rainfall during the summer of 2010–11. Despite this relief, the LMW icon site has experienced large-scale Lignum mortality over the last decade as a result of severe drought. There has been an increase in the number of Lignum plants recorded as dead or dormant since 2010–11 to more than 50% of the 450 plants originally surveyed. Some of the Lignum communities at LMW are located high on the floodplain and have not been inundated for more than 20 or 40 years.

Despite the widespread mortality, the majority of live Lignum plants surveyed were in good or moderate condition. It is anticipated that a transition to the recommended methodology under the TLM refinements program will increase our understanding of the condition of Lignum communities across the LMW icon site in future monitoring years.

It is recommended that additional investigations of Lignum condition and rootstock viability are undertaken at LMW to better understand the extent of mortality and the potential for recovery. These investigations could include piloting (with CSIRO) the use of remote sensing techniques to monitor the condition of Lignum-dominated communities across larger spatial scales.

Cumbungi

Interpretation of the objective *Increase diversity and abundance of wetland aquatic vegetation* with respect to Cumbungi led to development of the specific objective *Limit Cumbungi growth*. Limiting the abundance of Cumbungi allows other wetland aquatic vegetation to increase in diversity and abundance. The current level of cover is not yet considered detrimental to other wetland species and therefore the adopted objective *Limit Cumbungi growth* is being met. The Cumbungi population appears to be in the early stages of recovery. However there is no evidence yet that growth rates are reduced, or that populations have stabilised through effective management strategies to keep Cumbungi in check.

The spread of Cumbungi was prolific throughout all reaches within LMW over the monitoring periods from 2006–2011. In 2010–12, large-scale flooding and above-average rainfall ended the millennium drought (mid-1990s –2009). Flooding all but eliminated Cumbungi from the LMW Islands with the exception of Lower Wallpolla Creek. Cumbungi was not detected for two years post-flood at any site except Lower Wallpolla Creek.

In the current monitoring year (2015–16), Cumbungi was again found to be present in all reaches of the LMW icon site. While this re-colonisation is large-scale across the icon site, the percent cover is far below the degree that it was before flooding (2010–11).

Fish

Most of the metrics used to measure native fish condition at LMW returned positive results across all macrohabitats for 2015–16. This is primarily associated with the increase in the abundance and diversity of native small-bodied species. These results indicate that the adopted objective for fish at LMW *Maintain native fish populations, their relative abundance and diversity* is being met.

In terms of raw abundance, 2015–16 produced the highest catch of fish at LMW over the span of the TLM Condition Monitoring Program. For comparison, the next largest catch was experienced in 2010–11 (flood year). The catch during the flood year was dominated predominately by non-native species (~90%), particularly Common carp (*Cyprinus* carpio) and Eastern mosquitofish (*Gambusia holbrooki*), whereas native species dominated this year's catch (~90%).

Murray cod numbers have been increasing each year since the floods in 2010 and 2011, suggesting that successful spawning has led to subsequent recruitment, with the majority of individuals sampled ranging from 100 to 300 mm total length (young-of-year to 2+ and 3+ years old). Golden perch catch numbers have increased but are dominated by adult fish, with no signs of recruitment in 2015–16.

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 Condition Monitoring Program 2006–07 to 2015–16. This work was conducted by The Murray–Darling Freshwater Research Centre (MDFRC) and La Trobe University for the Mallee Catchment Management Authority (MCMA) and represents a deliverable requirement for Contract No. 15.1397 between La Trobe University and MCMA.

1.2 Report structure

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

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

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

1.3 Lindsay, Mulcra and 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: Lindsay, Mulcra and Wallpolla Islands, covers 26 156 ha in north-west Victoria, downstream of the Murray–Darling junction at Wentworth. The Lindsay–Mulcra–Wallpolla 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 the Lindsay–Mulcra–Wallpolla Islands is provided in Chapter 5 of The Living Murray Foundation Report (MDBC 2005).

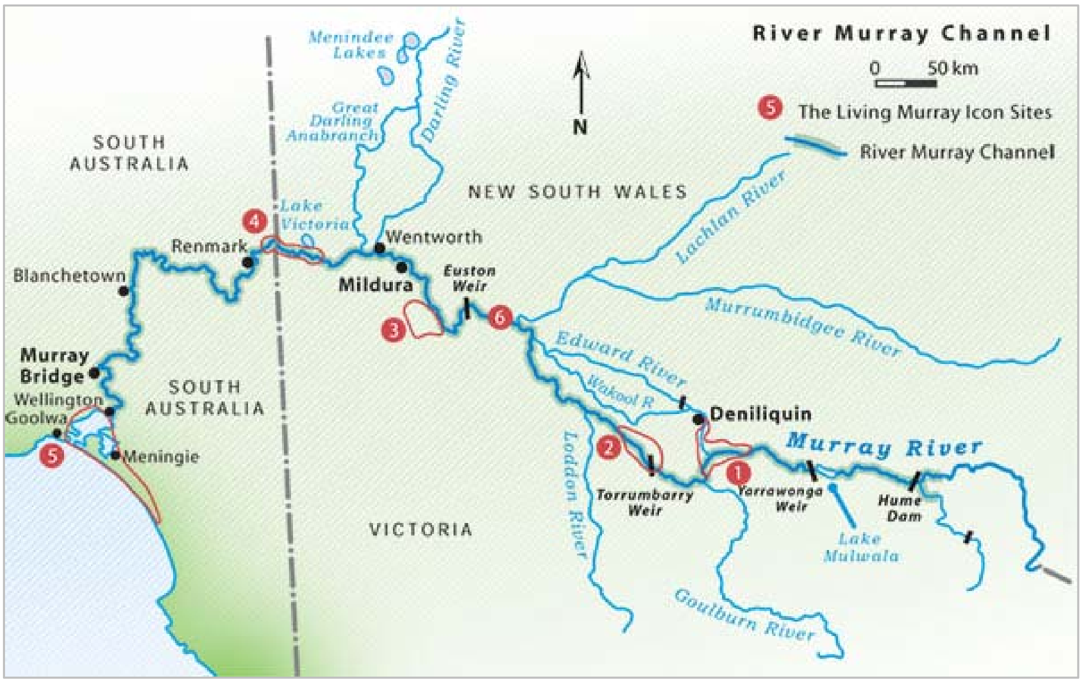


Figure 1-1. Locations of The Living Murray icon sites: (1) Barmah–Millewa Forest, (2) Gunbower–Koondrook–Perricoota Forest, (3) Hattah Lakes, (4) Chowilla Floodplain and Lindsay–Mulcra–Wallpolla Islands, (5) Lower Lakes, Coorong and Murray Mouth and (6) Murray River channel (image courtesy of MDBA).

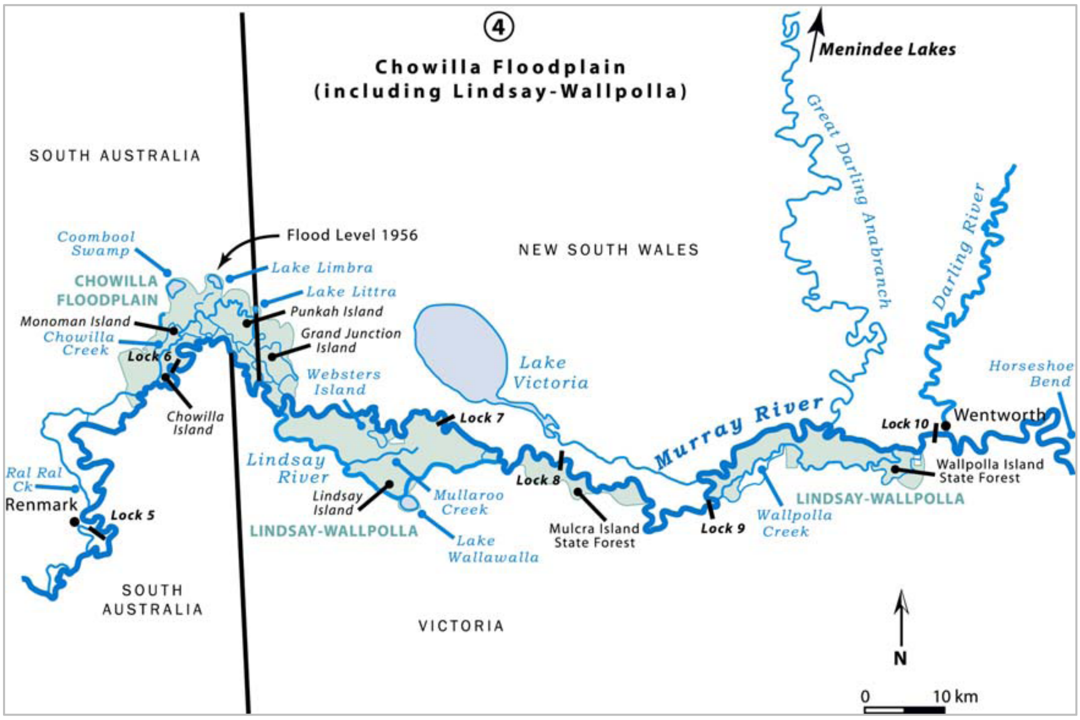


Figure 1-2 Chowilla Floodplain and LMW icon site (image courtesy of MDBA).

1.4 Hydrology

Drought conditions prevailed at LMW 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 that fell throughout the Murray–Darling Basin caused high flows in the both Murray and Darling Rivers. Flows at Lock 9 peaked at 74 500 ML.day^{-1} in February 2011 causing large areas of the LMW floodplain to become inundated. Flows subsided in the ensuing months before steadily rising again to 49 200 ML.day^{-1} in April 2012. They receded then peaked again slightly lower at 48 000 ML.day^{-1} in August 2012. Flows returned to base levels early in 2013, following which there have been only minor fluctuations in flow and river level.

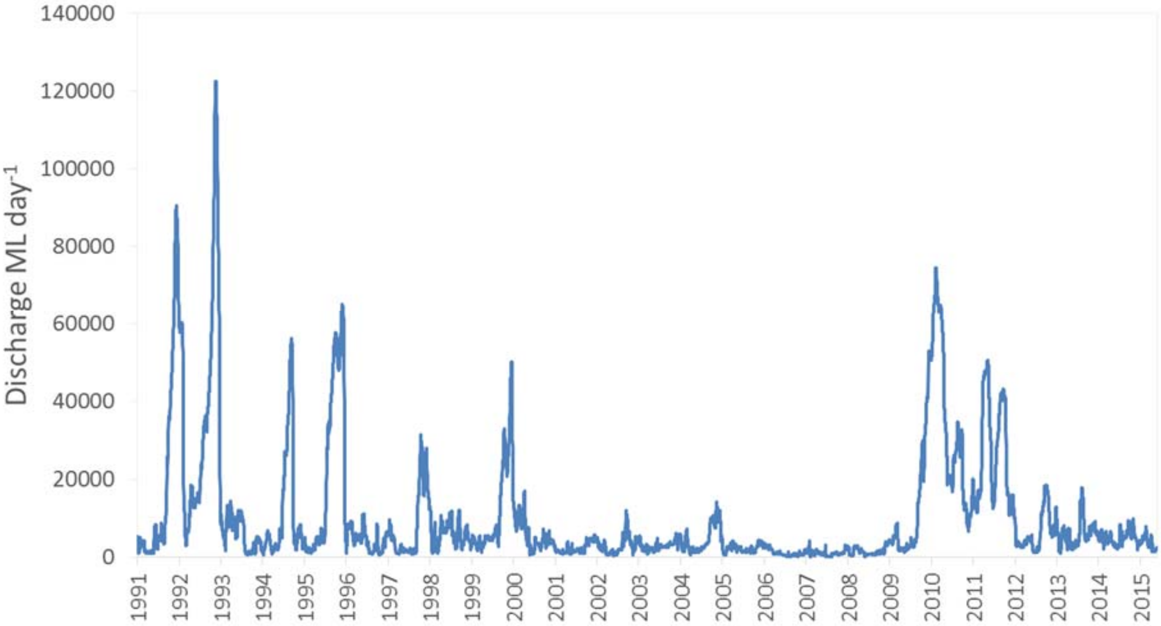


Figure 1-3. Mean daily discharge in the Murray River at Lock 9 for the period January 1992 to May 2016 (data courtesy of MDBA).

2 River Red Gum

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, River Red Gums play an important functional role in the floodplain ecosystem by means of contribution to 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 LMW.

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 River 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 River Red Gum communities at LMW through environmental works and the delivery of environmental water.

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

2.2 Ecological objectives

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

2.3 Methods

In order to address objectives relating to River Red Gum trees, two methods were employed: (i) tree condition monitoring and (ii) population size-class distribution assessments. Both components were evaluated at the community level (i.e. sites are randomly stratified within two River Red Gum water regime classes (WRCs); Table 2.1).

Comprehensive details on tree condition monitoring and size-class distribution methods are available in the Condition Monitoring Program design for the Chowilla Floodplain and the Lindsay, Mulcra and Wallpolla Islands (MDFRC 2011).

Table 2.1 The water regime classes used to define River Red Gum communities at LMW Islands. The WRCs are based on ecological vegetation classes and hydrological association (Ecological Associates 2007).

Water regime class	Component ecological vegetation classes	Characteristics
Red Gum forest	106 Riverine grassy forest 811 Grassy riverine forest	Found only in areas subject to the most frequent flooding regimes. This water regime class is subject to inundation in nearly all years, and winter floods maintain the density, size and health of River Red Gum.
Red Gum woodland	813 Intermittent swampy woodland 818 Shrubby riverine woodland 295 Riverine grassy woodland 815 Riverine swampy woodland	Occurs mainly in floodplain areas immediately surrounding wetlands and along water courses that are inundated by peaks in river flow during most years.

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 30 River Red Gum trees, were established in 2007–08 and sampled annually to 2015–16 (with the exception of 2010–11 due to flooding and 2014–15 when the program did not run).

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

2.3.2 Population demographics

Population size-class distribution surveys are used to inform population status assessments and, in conjunction with tree condition monitoring (mortality data), to inform population growth assessments. These assessments are used to evaluate the long-term sustainability of River Red Gum at LMW Islands and relate closely to the objective of maintaining extent (MDBA 2012b).

The size-class distribution of River Red Gum is assessed on a three-year rolling cycle such that for each year approximately one third of sites are sampled. Transects were established in 2006–07, 2007–08 and 2008–09, covering 57.3 ha, which represents approximately 1.7% of the extent of River Red Gum at LMW Islands.

Each transect was navigated end-to-end using a hand-held GPS. The DBH of each River Red Gum tree within each transect 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 for age where it is assumed that, on average, the larger the DBH of the tree, the older it is.

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

2.4 Indices and points of reference

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

2.4.1 Tree condition

Reference points for River Red Gum condition at LMW Islands are currently in development. As an interim approach for 2015–16 reporting, the target used for River Red Gum condition at LMW was:

- *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 (Table 2.2). This target is based on TLM condition monitoring data collected in 2007–08 to 2012–13 that indicates that River Red Gum trees with less than 40% foliated crown are at significantly higher risk of mortality than trees with more foliated crowns (MDFRC, unpublished data).

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

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

2.4.2 Population demographics

Two methods for evaluating population status were used:

- population status index
- population growth index.

Population status

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

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

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

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

Population growth

The population growth index was based on the net population growth for Red Gum forest. This was calculated as the difference between the running three-year averages for recruitment and mortality. The approach for calculating recruitment was based on the current understanding of growth rates and estimated age at maturity and was calculated on population size-class data. Two assumptions are made: (i) River Red Gum at LMW grow in diameter at approximately 1 cm per year (Henderson 2010) and (ii) trees mature at approximately 10 years of age (George 2004). These assumptions are based on information from previous studies and, although the data is not comprehensive, they are considered the most reliable available. Based on these assumptions, annual recruitment into the adult population may be measured as the number of trees that, when monitored once each year, record a DBH between 10 and 11 cm. To apply this approach to a population that is monitored on a three-year rolling basis (one third of sites monitored each year; each site revisited once every three years), annual recruitment into the adult population was calculated as one third of the number of trees between 10 and 13 cm DBH.

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

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

2.5 Results

2.5.1 Tree condition

For 2015–16, the target of 85% of River Red Gum trees exceeding a crown extent score of ≥ 4 was met for both water regime classes at LMW.

Both water regime classes—Red Gum forest and Red Gum woodland—at LMW have followed similar trajectories for tree condition. During 2008–09, both water regime classes had the lowest recorded mean frequency of trees with a crown extent score ≥ 4 (Figure 2-1 and Figure 2-2). The following year (2009–10), mean frequency increased, but was still below the 85% target value. A further increase in mean frequency occurred to 2011–12; at this time the 85% target was exceeded. Since 2011–12 mean frequency has exceeded the 85% target value and has varied very little for both water regime classes.

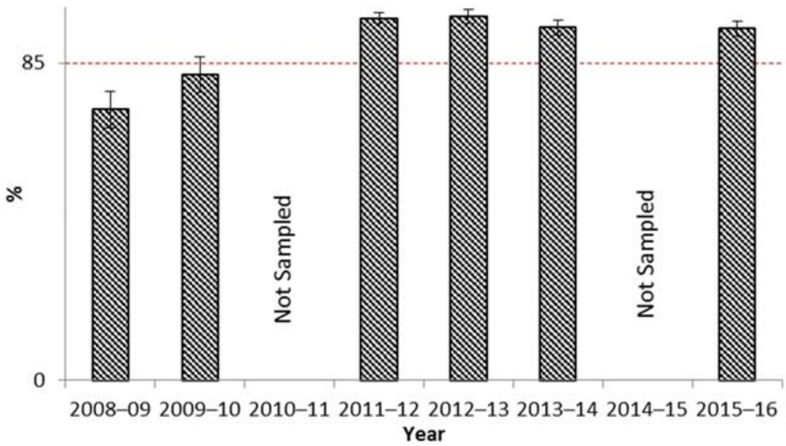


Figure 2-1. Mean frequency (± SE) of trees with crown extent scores ≥ 4 recorded for Red Gum forest at sites sampled annually between 2008–09 and 2015–16 (except for 2010–11 when flooding prevented access and 2014–15 when the program did not run).

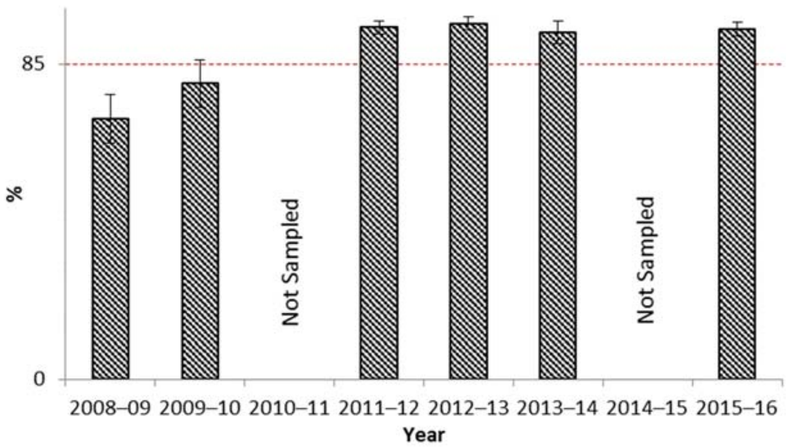


Figure 2-2. Mean frequency (± SE) of trees with crown extent scores ≥ 4 recorded for Red Gum woodland at sites sampled annually between 2008–09 and 2015–16 (except for 2010–11 when flooding prevented access and 2014–15 when the program did not run).

2.5.2 *Population demographics*

Population status

Size-class frequency distributions for Red Gum forest for 2012–13 and 2015–16 were similar, with comparatively large numbers of 0–15-cm DBH trees evident (Figure 2-3). Recent germination of seedlings (0–2 cm DBH) was observed in 2012–13. Many of these evidently survived and developed, as evidenced by a shift in the size-class distribution towards 1–4-cm DBH trees in 2015–16 (Figure 2-4). Recent germination of River Red Gum was also evident in 2015–16.

The River Red Gum woodland population was dominated by 0–15-cm DBH trees in 2015–16, however not to the same extent as during 2012–13 (Figure 2-6). A four-and-a-half-fold decline in the total number of 0–15-cm DBH trees occurred between 2012–13 and 2015–16.

The population status index score for both River Red Gum water regime classes, after showing marginal improvements in the mean between 2006–07 and 2012–13, has not improved further to 2015–16 (Figure 2-5 and Figure 2-8), however none of the changes would be considered statistically significant.

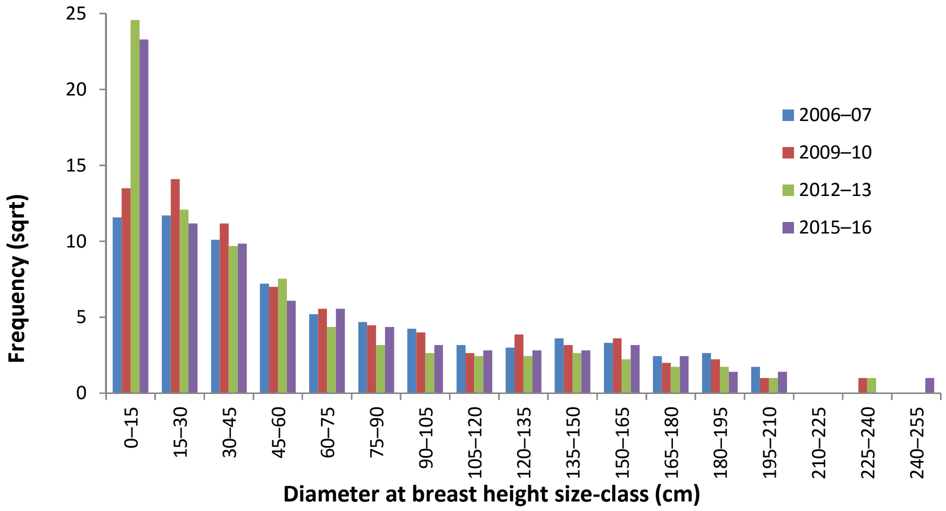


Figure 2-3. Size-class distribution of live River Red Gum forest trees (0–255 cm DBH) at LMW; n(2006–07) = 551, n(2009–10) = 678, n(2012–13) = 969, n(2015–16) = 907.

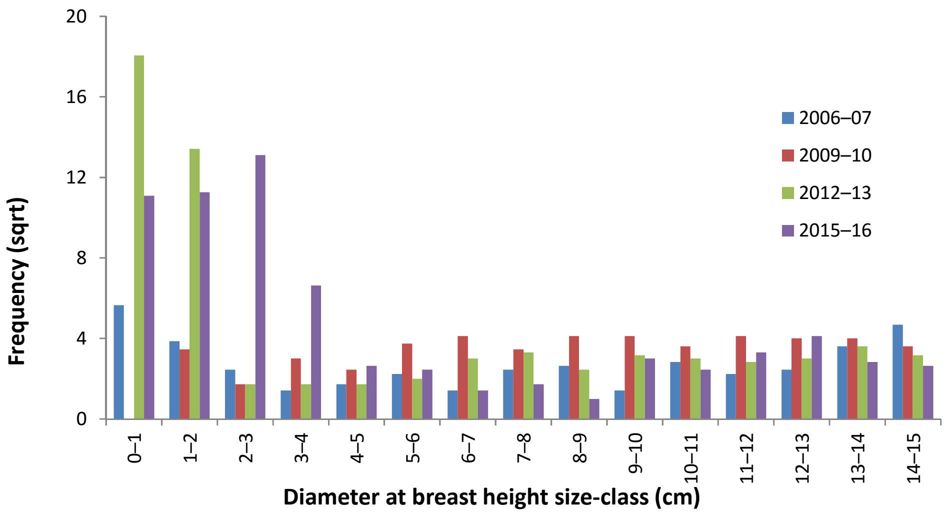


Figure 2-4. Size-class distribution of live River Red Gum forest trees (0–15 cm DBH) at LMW; n(2006–07) = 134, n(2009–10) = 182, n(2012–13) = 604, n(2015–16) = 543.

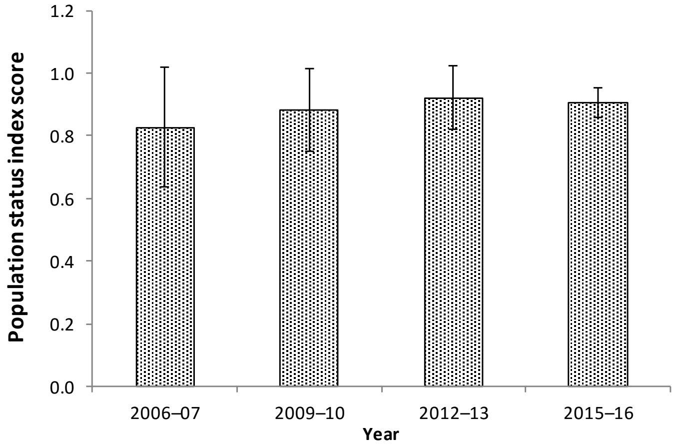


Figure 2-5. Population status index (± 95% CI) for Red Gum forest at LMW, calculated based on level of correlation with the reference ‘inverse J-shaped’ curve.

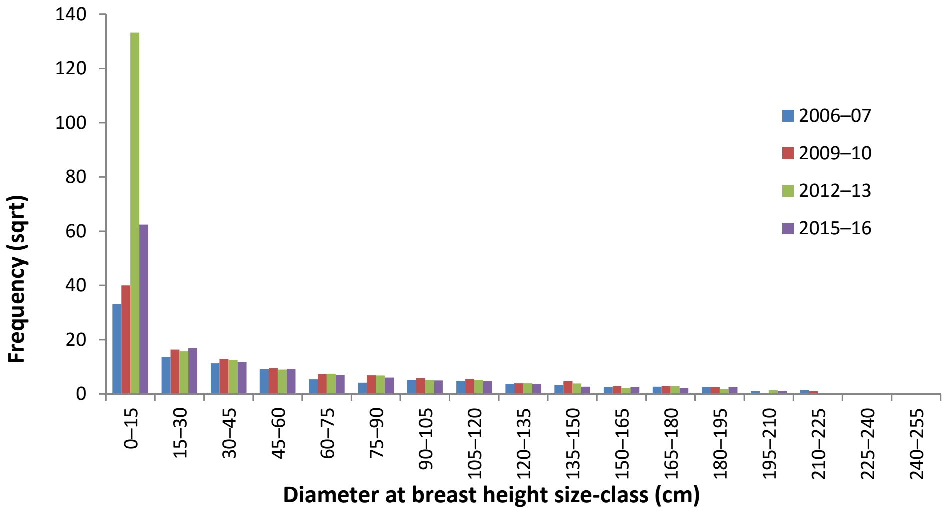


Figure 2-6. Size-class distribution of live River Red Gum woodland trees (0–255 cm DBH) at LMW; n(2006–07) = 1640, n(2009–10) = 2356, n(2012–13) = 18 450, n(2015–16) = 4590.

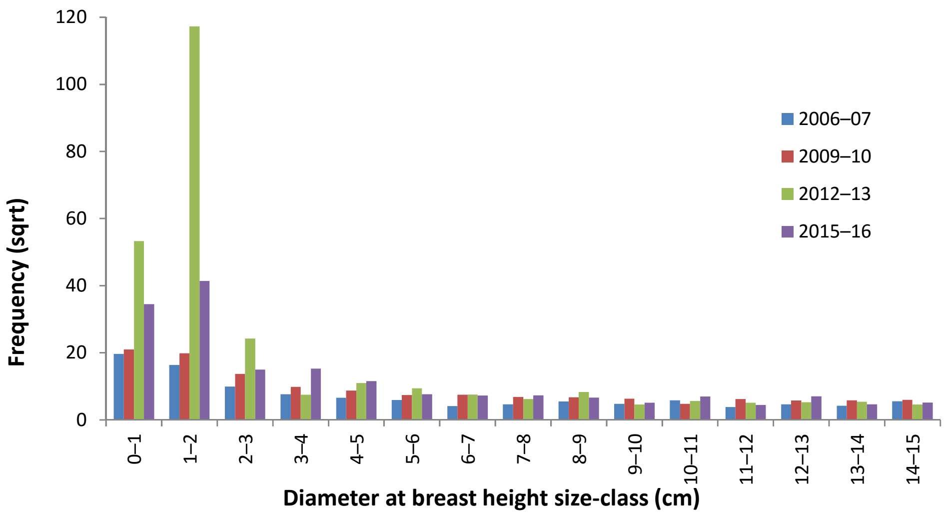


Figure 2-7. Size-class distribution of live River Red Gum woodland trees (0–15 cm DBH) at LMW; n(2006–07) = 1101, n(2009–10) = 1605, n(2012–13) = 17 758, n(2015–16) = 3902.

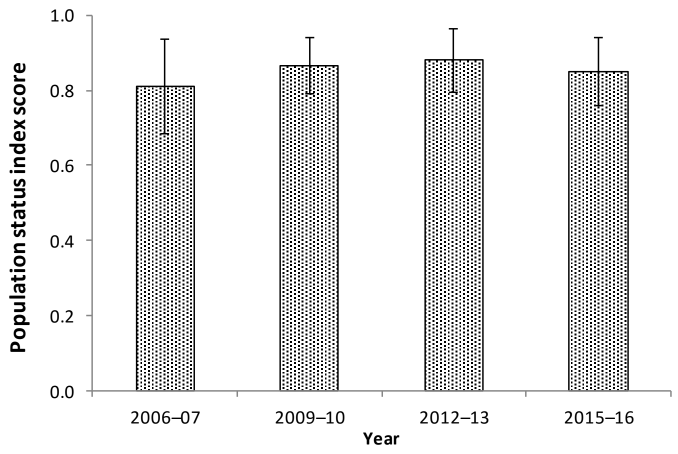


Figure 2-8. Population status index (±95% CI) for Red Gum woodland, calculated based on level of correlation with the reference 'inverse J-shaped' curve.

Population growth

There has been a net population gain for all years for both River Red Gum water regime classes at LMW (Figure 2-9 and Figure 2-10), which indicates that recruitment has exceeded mortality. Net gains for Red Gum forest and Red Gum woodland were 4.45% and 2.25% respectively for 2015–16. Mortality for Red Gum forest has been relatively stable, while recruitment has been more variable. For Red Gum woodland there has been a trend of declining mortality since 2008–09.

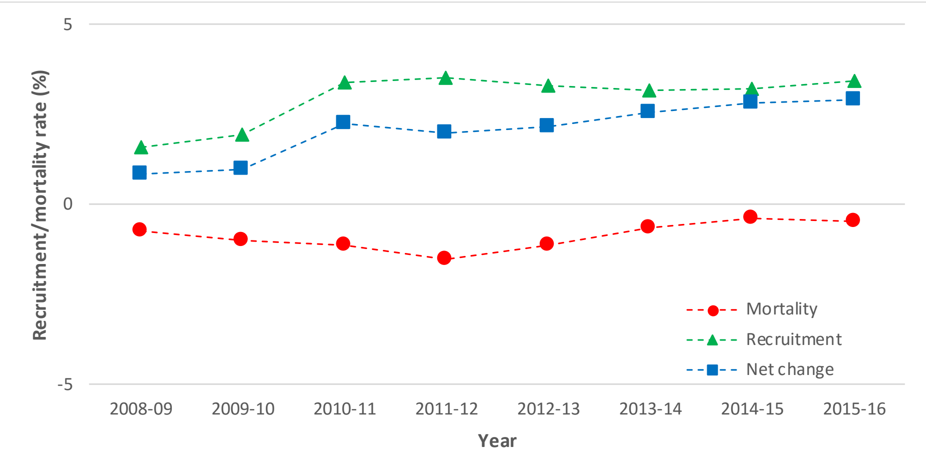


Figure 2-9. Annual recruitment, mortality and net population change for Red Gum forest at LMW.

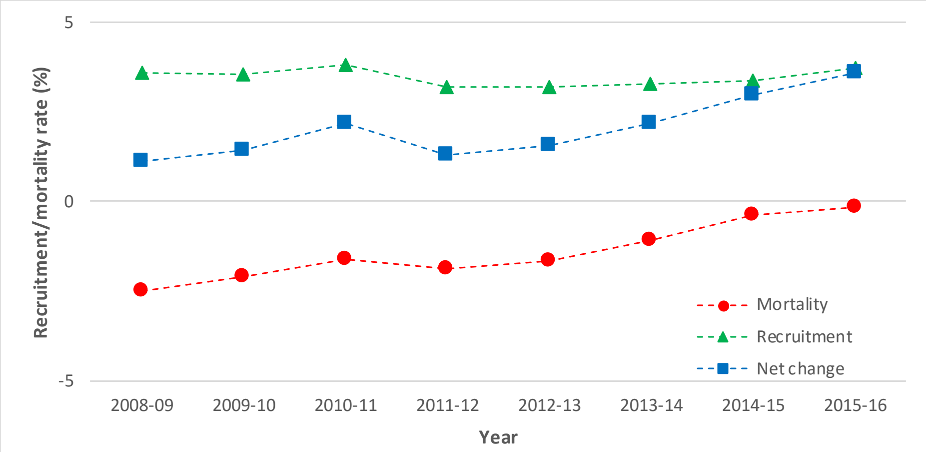


Figure 2-10. Annual recruitment, mortality and net population change for Red Gum woodland at LMW.

2.6 Discussion

River Red Gum condition in both WRCs at LMW showed a general improvement from 2008–09 to 2011–12. This time period coincided with higher-than-average rainfall, particularly in 2010–11, and flooding in 2010–11 which would have influenced proportions of both water regime classes. River Red Gum respond positively to increased water availability with an increase in canopy extent and density. This does depend on the tree's initial health, with trees in poor condition generally not responding, or taking longer to do so than trees in comparatively better health. The increased water availability during 2010–11 would also have recharged and raised the water table, making it more accessible to trees like River Red Gum and helping them maintain condition for a prolonged period of time. This is perhaps one of the reasons for the relatively stable maintenance of tree condition since 2011–12.

The populations of both River Red Gum WRCs have been dominated by seedlings since 2012–13. Seedling germination prior to 2012–13 was prompted by flooding in 2010–11. In 2015–16, a decline in the proportion of 0–15-cm DBH trees occurred, predominantly due to a loss in trees with < 5 cm DBH as seedlings from this earlier germination perished. Increased competition with adjacent seedlings due to adverse weather conditions (i.e. lack of further flooding, below-average rainfall) and natural attrition are the likely causes of death. While the ongoing germination of seedlings at LMW is an important step in maintaining the River Red Gum population, the survival of a proportion of these trees to maturity is imperative to the long-term survival of this species.

Where recruitment equals or exceeds mortality it is deemed that a viable population exists. An annual net population gain for the River Red Gum community at LMW for all years since 2008–09 meets this requirement.

As of 2015–16, it is deemed that the overall health of River Red Gum is being maintained above current targets; therefore the ecological objective *Maintain current condition and extent of River Red Gum communities to sustain species assemblages and processes typical of such woodland* at the LMW icon site is being met.

3 Black Box

DAVID WOOD

3.1 Introduction

Black Box (*Eucalyptus largiflorens*) is a common flora species at the LMW Islands, and one of only a few large tree species. Black Box generally occur higher on the floodplain (i.e. in less frequently inundated environments) 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).

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 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 onto natural high flows.

Condition monitoring reports on Black Box condition and population status at the icon-site scale. Black Box are monitored on an annual basis as outlined in the Condition Monitoring Program design for Chowilla Floodplain and the Lindsay, Mulcra and Wallpolla Islands (MDFRC 2011). Monitoring is specifically tailored to determine if the ecological objective for Black Box is being met.

3.2 Ecological objectives

The ecological objective for Black Box at LMW is (MDBA 2012b):

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

3.3 Methods

In order to address objectives relating to Black Box trees two methods were employed: (i) tree condition monitoring and (ii) population size-class distribution assessments. Both components were evaluated at the community scale (i.e. sites are randomly stratified within two Black Box WRCs;

Table 3.1).

Comprehensive detail on tree condition monitoring and size-class distribution assessments is provided in the Condition Monitoring Program design for the Chowilla Floodplain and the Lindsay, Mulcra and Wallpolla Islands (MDFRC 2011).

Table 3.1. The WRCs used to define Black Box communities at LMW based on hydrological association (Ecological Associates 2007).

Water regime class	Component ecological vegetation classes	Characteristics
Black Box swampy woodland	823 Lignum swampy woodland	Woodland commonly associated with Lignum understorey. Generally occurs at lower terraces and thus more prone to flooding than riverine chenopod woodland.
Riverine chenopod woodland	103 Riverine chenopod woodland	Woodland on most elevated riverine terraces. Naturally subject to only extremely infrequent incidental shallow flooding from major events if flooded at all.

3.3.1 Tree condition

Tree condition monitoring is a ground-based monitoring method used to detect changes in Black Box condition based on assessing a number of variables for each tree (MDBA 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 DBH was measured (MDBA 2012a).

Twenty-seven sites each comprised of 30 Black Box trees were established in 2007–08 and sampled annually to 2015–16, except for 2010–11 due to flooding and 2014–15 when the program did not run.

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

3.3.2 Population demographics

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

Size-class distribution of Black Box is assessed on a three-year rolling cycle such that for each year approximately one third of sites are sampled. Transects were established in 2006–07, 2007–08 and 2008–09, covering 96.6 ha, which represents approximately 1.64% of the extent of Black Box at LMW.

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

Sites first surveyed in 2006–07 and reassessed in 2009–10, 2012–13 and 2015-16 are presented in this report. To examine temporal trends in population structure, all live trees were grouped into 15-cm DBH categories.

3.4 Indices and points of reference

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

3.4.1 Tree condition

Reference points for Black Box condition at LMW are currently in development. As an interim approach for 2015–16 reporting, the target used for Black Box condition at Mulcra was used (Henderson et al 2014):

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

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

3.4.2 Population demographics

Two methods for evaluating population status were used:

- population status index
- population growth index.

Population status

The population status index was calculated as described in section 2.4.2.

Population growth

The population growth index was calculated as described in section 2.4.2. Growth rates were based on Black Box growth rate data that show that, on average, Black Box trees at LMW grow at a rate of approximately 1 cm per year (MDFRC, unpublished data) and on the assumption that trees mature at approximately 10 years of age (George 2004).

3.5 Results

3.5.1 Tree condition

Black Box from both WRCs (Black Box swampy woodland and riverine chenopod woodland) at LMW have followed a similar trend in condition (Figure 3-1 and Figure 3-2). An improvement in the mean frequency of trees with a crown extent score ≥ 4 occurred from 2008–09 to 2011–12. Since 2011–12 mean frequency has generally remained steady.

In the period 2011–12 to 2015–16, the mean frequency of trees in Black Box swampy woodland exceeded the target value of 80% of crown extent scores ≥ 4. However, we cannot say with confidence that the target has been met as the 95% confidence intervals falls beyond the 80% target.

A similar trend has occurred in riverine chenopod woodland, with the mean frequency of crown extent scores ≥ 4 exceeding the target value of 80% in the period 2011–12 to 2015–16.

Overall, for 2015–16, the target of 80% of Black Box trees exceeding a crown extent score of ≥ 4 was met for both WRCs at LMW.

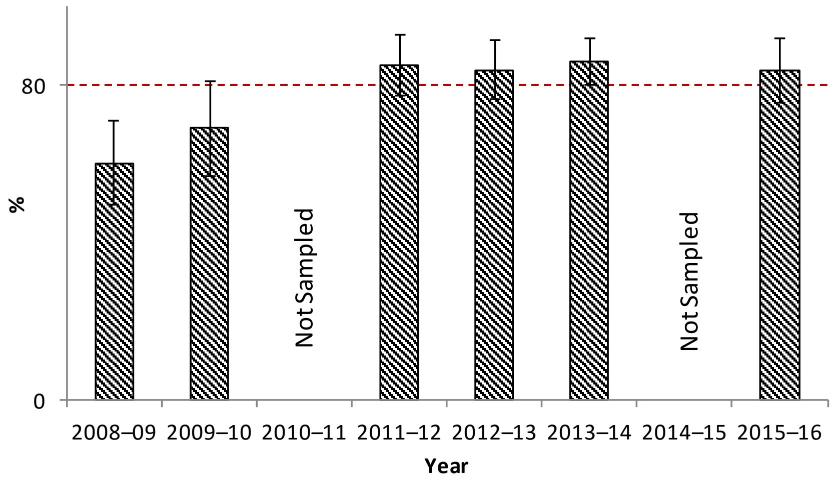


Figure 3-1. Mean frequency (± SE) of trees with crown extent scores ≥ 4 recorded for Black Box swampy woodland at sites sampled annually in summer between 2008–09 and 2015–16, except for 2010–11 when flooding prevented access and 2014–15 when the program did not run.

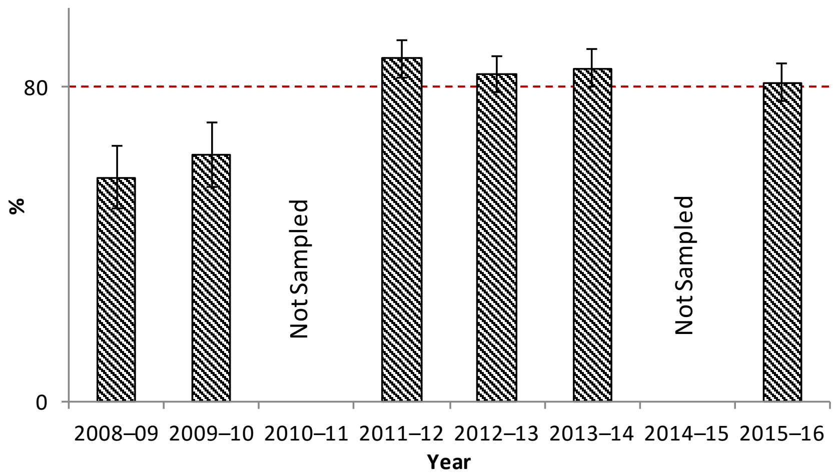


Figure 3-2. Mean frequency (± SE) of trees with crown extent scores ≥ 4 recorded for riverine chenopod woodland at sites sampled annually in summer between 2008–09 and 2015–16, except for 2010–11 when flooding prevented access and 2014–15 when the program did not run.

3.5.2 Population demographics

Population status

The size-class frequency distribution for both Black Box swampy woodland and riverine chenopod woodland, over the previous three monitoring periods, shows a population dominated by 15–30-cm DBH trees (Figure 3-3 and Figure 3-6). For Black Box swampy woodland there was an overall decline in the total number of trees over the last three monitoring periods, particularly between 2012–13 and 2015–16. This decline was primarily in the 0–15-cm DBH size-class trees (Figure 3-4). The same has occurred for riverine chenopod woodland, with an overall decline in the number of trees, most evident from the 0–15-cm DBH size-class (Figure 3-8).

After showing some improvement in mean population status index score between 2009–10 and 2012–13, the population status index score for Black Box swampy woodland did not change substantially to 2015–16. A large variability exists within the Black Box swampy woodland population, as indicated by broad confidence intervals (Figure 3-5). Overall, the riverine chenopod woodland population has remained relatively unchanged in terms of its likeness to an ideal, J-shaped curve across all sampling periods (Figure 3-8).

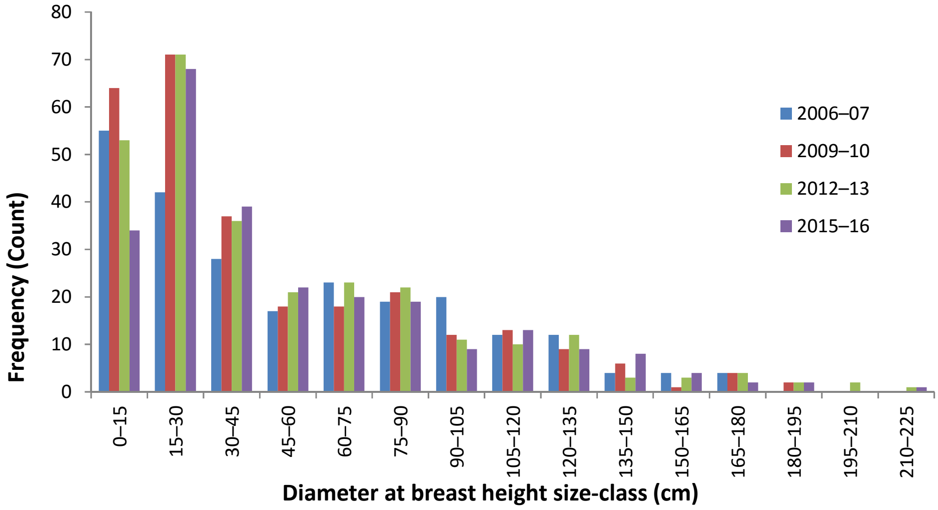


Figure 3-3. Size-class distribution of live Black Box swampy woodland trees (0–225 cm DBH) at LMW; n(2006–07) = 240, n(2009–10) = 276, n(2012–13) = 274, n(2015–16) = 250.

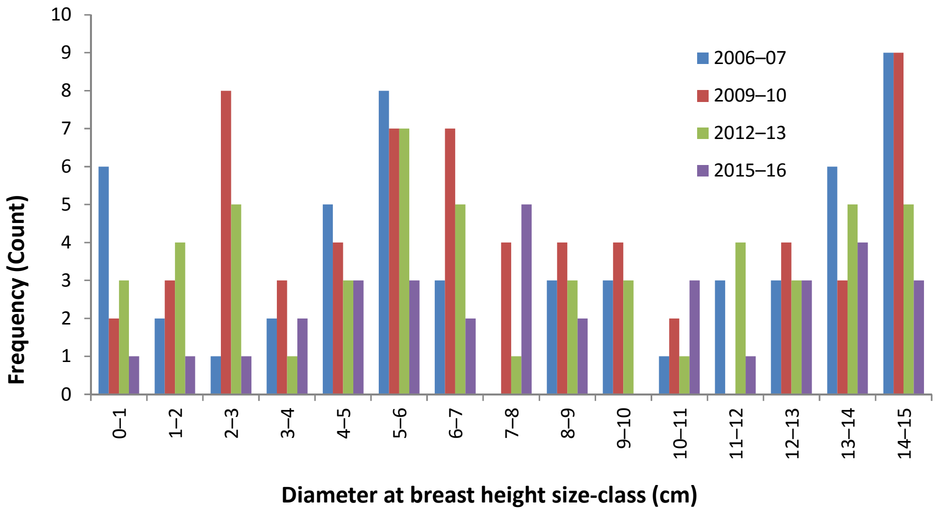


Figure 3-4. Size-class distribution of live Black Box swampy woodland trees (0–15 cm DBH) at LMW; n(2006–07) = 55, n(2009–10) = 64, n(2012–13) = 53, n(2015–16) = 34.

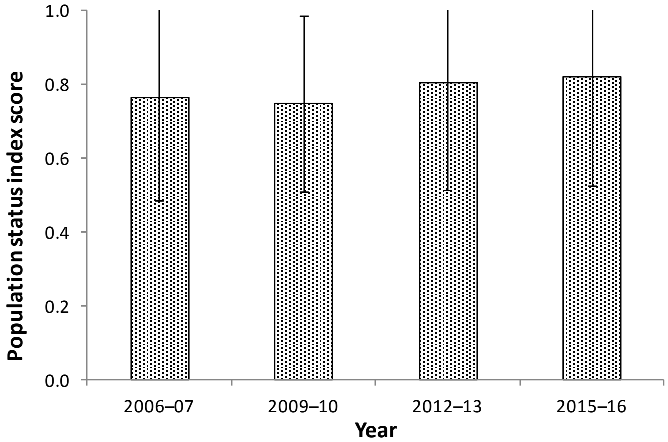


Figure 3-5. Population status index (± 95% CI) for Black Box swampy woodland, calculated based on the level of correlation with the reference 'inverse J-shaped' curve.

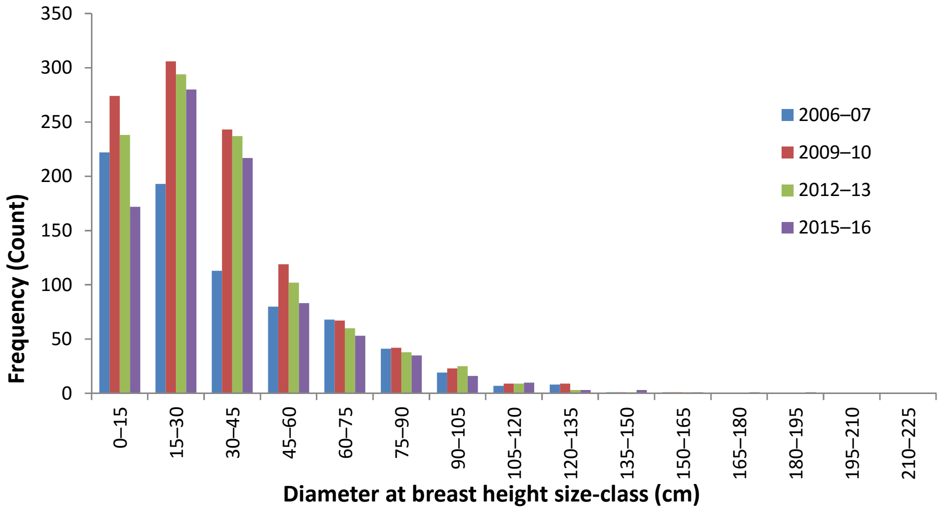


Figure 3-6. Size-class distribution of live riverine chenopod woodland Black Box trees (0–225 cm DBH) at LMW; n(2006–07) = 753, n(2009–10) = 1094, n(2012–13) = 1008, n(2015–16) = 875.

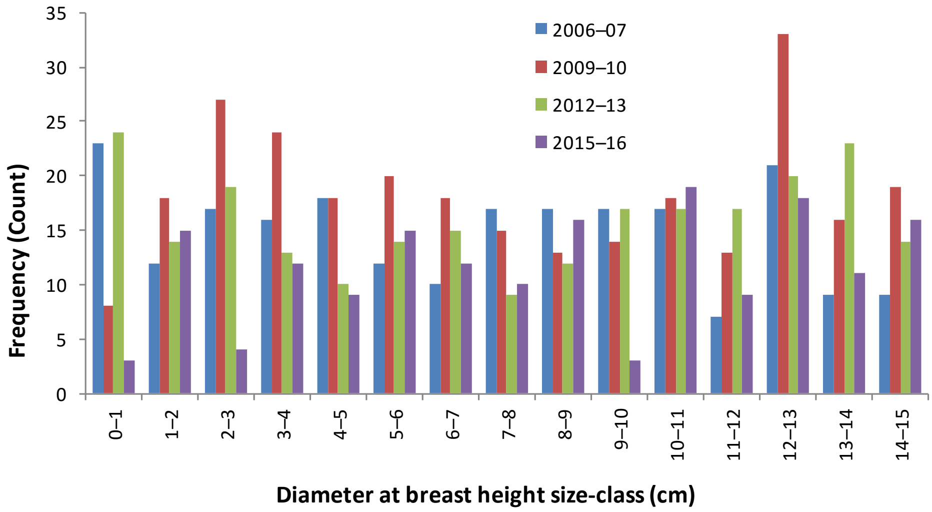


Figure 3-7. Size-class distribution of live river riverine chenopod woodland Black Box trees (0–15 cm DBH) at LMW; n(2006–07) = 222, n(2009–10) = 274, n(2012–13) = 238, n(2015–16) = 172.

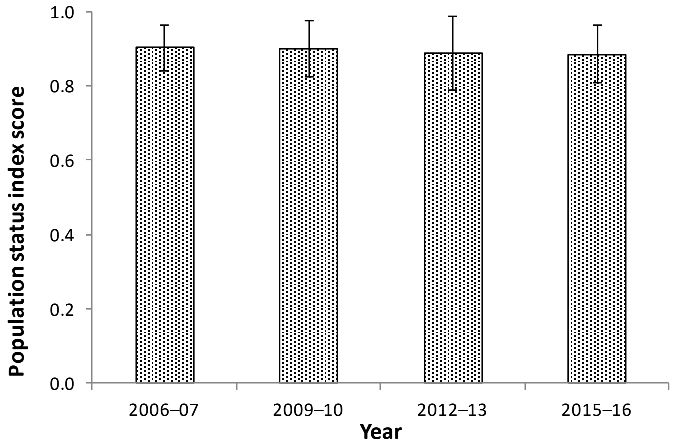


Figure 3-8. Population status index (± 95% CI) for riverine chenopod woodland Black Box trees, calculated based on level of correlation with the reference 'inverse J-shaped' curve.

Population growth

In 2015–16, marginal net population gains for riverine chenopod woodland and Black Box swampy woodland WRCs occurred (Figure 3-9 and Figure 3-10). Between 2008–09 and 2013–14 the Black Box swampy woodland population experienced a net decline, which stabilised in recent years (2014–16). The riverine chenopod woodland WRC has recorded a net annual population gain since 2008–09.

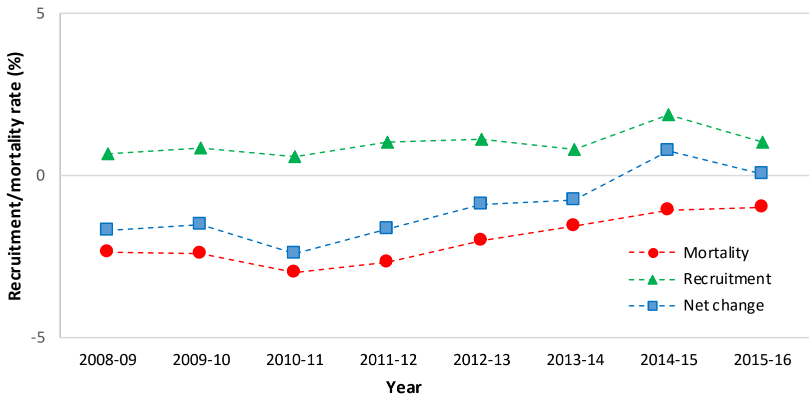


Figure 3-9. Annual recruitment, mortality and net population change for Black Box swampy woodland at LMW.

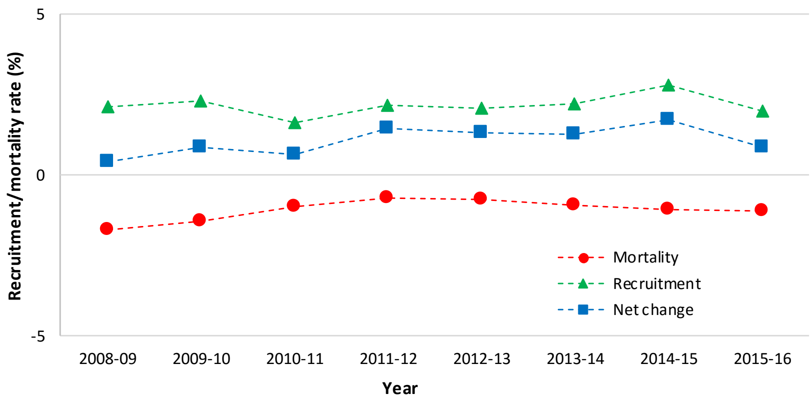


Figure 3-10. Annual recruitment, mortality and net population change for Black Box in riverine chenopod woodland at LMW.

3.6 Discussion

The current condition of Black Box at LMW is of some concern. While there was improvement in tree condition between 2008–09 and 2011–12, it appears that this has plateaued. The initial improvement was likely associated with above average rainfall during 2010–11, which caused localised flooding. Wide-scale flooding also occurred at that time but would have had very little direct influence on much of the Black Box community, particularly the riverine chenopod woodland WRC as it is located on the highest points on the floodplain. Increased water availability during this time would have recharged groundwater reserves. Black Box are thought to access groundwater when available soil moisture disappears (Jolly & Walker 1996). Since 2011–12, there has been no flooding (with the exception of some environmental watering at Mulcra in 2013 and 2014) and below-average rainfall (particularly during 2012, 2013 and 2015) (BOM 2016). It appears that reduced water availability is starting to result in a downturn in Black Box condition, particularly for the riverine chenopod woodland WRC.

Germination of Black Box has been low for both WRCs across all sampling periods. Black Box germination tends to be sporadic, with higher instances generally recorded following flooding or heavy rain (George 2004). The vast majority of Black Box at LMW has not been inundated for a considerable period of time (> 20 years). For populations of Black Box to be sustained at LMW future large-scale germination events, with survival of a proportion of seedlings to mature trees, are critical.

Seedling survival in recent years is an issue at LMW. Adverse weather (e.g. the millennium drought and, more recently, below-average rainfall) and lack of flooding are likely contributing to the death of smaller individuals. Smaller trees with less-well-developed root systems may still rely on local rainfall and flooding for survival, while larger mature trees are able to access and utilise groundwater (Roberts & Marston 2011).

Where recruitment equals or exceeds mortality it is deemed that a viable population exists. For the Black Box swampy woodland population, net population gain has occurred annually since 2008–09, indicating a viable population. The riverine chenopod woodland population is of some concern as it has experienced a long trend of annual population loss. This is likely due to a lack of flooding. Flooding is known to increase seedling germination and to assist with seedling growth and survival to recruitment. Only recently (since 2014), the trend of annual population loss has been halted. Recovery of the population may take several consecutive years of net growth. The overall population of Black Box at LMW is of concern and needs to be managed and monitored closely to prevent further decline.

As of 2015–16, although Black Box condition currently meets targets, the overall health of the population is in decline therefore the ecological objective *Improve condition to sustain species assemblages and processes typical of Black Box woodland* at the LMW icon site is not being met.

4 Wetland vegetation communities

FIONA FREESTONE AND CHERIE CAMPBELL

4.1 Introduction

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

4.2 Ecological objectives

The vision for the 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 is:

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

The Living Murray program is currently undergoing refinements that involve the development of operational objectives and the identification of suitable ecological indices that link back to the vision statement and ecological objective (Brown et al. 2015; Robinson 2014a, b). At the time this report was compiled, operational objectives and indices for wetland vegetation were not yet developed. As an interim measure for this monitoring period, this chapter reports on the established vision statement and overarching ecological objective by examining changes in wetland vegetation over time. This was achieved by analysing species richness, non-native species (i.e. weediness) and changes in the community composition of wetland vegetation. This information may be used to assist in developing indices, or points of reference, for future condition monitoring reporting.

4.3 Methods

This section summarises the methods used to monitor and analyse species richness, non-native species and changes in wetland community composition over time. When condition monitoring began in 2007–08, wetland sites were established to represent the various sizes, shapes, commence-to-flow levels and vegetation communities that exist at LMW wetlands. Sites were selected using satellite imagery, Geographic Information System (GIS) models and layers, such as RiM-FIM (Overton et al. 2006), and local knowledge. Ten wetlands have been surveyed annually since 2007–08. Scotties Billabong was previously surveyed as an extension of a floodplain site (Lindsay Island floodplain site 2). This site has since demonstrated the ability to retain water following inundation, in a way that is more suited to the definition of a wetland than that of a floodplain. Data from 2007–08 to 2015–16 was collected using methods described in the floodplain chapter of this report, though it is proposed that this site be transitioned to wetland data collection methods over the coming years. In the meantime, Scotties Billabong was included in the wetland chapter for discussion in 2013–14 and 2015–16. In 2009–10, a new site was established and surveyed at Lake Wallawalla, bringing the total number of wetlands surveyed at LMW to 12.

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 treeline). For the number of quadrats and elevations at each individual wetland, refer to section 3.1 of Part B of this report. Details of the survey methods used at LMW can be found in section 6 of The Living Murray: Condition Monitoring Program design for Chowilla Floodplain and the Lindsay, Mulcra and Wallpolla Islands (MDFRC 2011). 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 (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. The seasonality of plant life cycles means that some species may not have been present at the time of the survey.

4.3.1 Wetland inundation state

The historic and current inundation state of each wetland provided context for data analysis. The hydrological state of each wetland in 2015–16 is given in Table 4.1. All wetlands were inundated during the most recent natural flood in 2010–11. In 2015–16, Websters Lagoon, Wetland 33 and the Mulcra Lower Horseshoe were dry for the first time during monitoring since the natural flood. Lake Wallawalla was drawing down after being filled with environmental water in spring 2015.

Table 4.1. Hydrological state of each wetland surveyed at LMW in 2015–16.

Key: long-dry = wetland was dry at the time of the survey and had not held water for more than two years previously; 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.

Island	Wetland	Hydrological state 2015–16	Environmental water delivered
Lindsay	Bilgoes Billabong	Long-dry	2005
	Bottom Island	Long-dry	NA^
	Crankhandle	Long-dry	2005; 2006; 2008; 2009; 2010
	Lake Wallawalla	Drawdown	2010; 2012; 2015
	Scotties Billabong	Long-dry	2005; 2006; 2008; 2009; 2010
	Upper Lindsay	Long-dry	NA
	Upper Mullaroo	Long-dry	2005; 2006; 2008; 2009; 2010
	Websters Lagoon	Intermittent-dry	2008; 2009; 2010
	Wetland 33	Intermittent-dry	NA
Mulcra	Mulcra Lower Horseshoe	Intermittent-dry	2005; 2006; 2008; 2009; 2013
	Mulcra Upper Horseshoe	Long-dry	2005; 2006; 2008; 2009; 2013
Wallpolla	Lilyponds	Long-dry	2005; 2006; 2008; 2009; 2010

^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. During this survey (2015–16), one quadrat was inundated to approximately 30 cm depth as water began trickling into the wetland.

4.3.2 Plant species classification

Plant species identification

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

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

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

Functional groups

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

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

Table 4.2. Functional groups (FG) used to classify plant species recorded in wetlands at LMW.

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

4.3.3 Data analysis

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

Table 4.3. Water regime class for each wetland at LMW.

Water regime class^	Wetlands
Semi-permanent wetlands	Lilyponds, Mulcra Lower Horseshoe, Mulcra Upper Horseshoe, Websters Lagoon
Ephemeral wetlands	Bilgoes Billabong, Bottom Island, Crankhandle, Scotties Billabongɨ, Upper Lindsay, Upper Mullaroo, Wallawalla, Wetland 33

^Information for WRC classification is largely based on the Mallee Catchment Management Authority (CMA) preliminary targets document that was developed for condition monitoring in 2012–13 and the Ecological Associates report (2007). The latter makes specific reference to Lilyponds and Lake Wallawalla in relation to associated WRCs. The remaining classifications are adapted from the Mallee CMA document. Ecological Associates (2007) make reference to Wetland 33 as not retaining water for long periods of time, however this wetland retained water from the natural flood (2010–11) for approximately three years and was dry for the first time in this monitoring year (2015–16). It is recommended that WRC classifications be reviewed for LMW wetlands.
ɨScotties Billabong has been added to the ephemeral wetlands classification based on local knowledge of the wetland.

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

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

4.4 Results

4.4.1 Species richness

A total of 109 plant species were recorded in 2015–16 across all wetlands at LMW (Table 4.4). The greatest species richness was recorded in 2009–10 during the drought. This is attributed to environmental watering that occurred at six wetlands in 2009. The lowest number of species was recorded in 2010–11 in association with natural flooding, when species from the floating functional group dominated the plant community. The total number of native and non-native species recorded in each year is displayed in Figure 4-1. For a comprehensive list of all plant species recorded at each site, refer to section 3 in Part B of this report.

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

Monitoring year	2007–08	2008–09	2009–10	2010–11	2011–12	2012–13	2013–14	2015–16
Number of sites surveyed	11	11	12	9^	12	12	12	12
Total species recorded	68	104	129	6	124	87	93	109
Total species recorded at original sites*	68	104	112	6	99	71	77	91

^Upper Mullaroo, Websters Lagoon and Wetland 33 were not surveyed in 2010–11 due to access issues associated with widespread flooding.
*Includes only the 11 sites that were originally surveyed in 2007–08, with the exception of 2010–11 when only nine sites were surveyed.

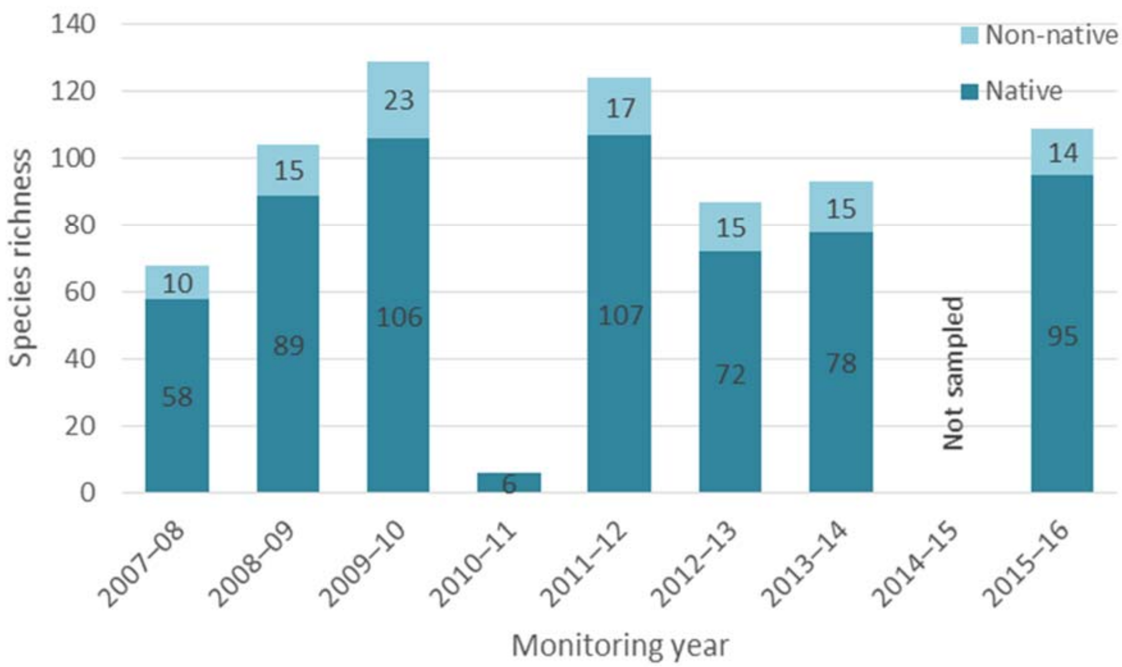


Figure 4-1. Number of native and non-native plant species recorded across all wetlands at LMW in all monitoring years (n = 11 in 2007–08 and 2008–09; n = 9 in 2010–11; n = 12 in all other years).

The most common wetland species recorded at LMW in 2015–16 are listed in Table 4.5. Common species are defined as those with total abundance (number of survey quadrats in which the species occurred) of greater than 100 (out of a possible 2520, being the number of 1 m x 1 m cells surveyed across all wetlands).

Table 4.5. The most common plant species recorded in wetlands at the LMW icon site in 2015–16.

Key: Atl = amphibious monocotyledons, Tda = terrestrial plants preferring damp habitats, Tdr = drought-tolerant species.

Scientific name	Common name	Functional group	Abundance (out of 2520)
Centipeda cunninghamii	Common Sneezeweed	Atl	332
Senecio cunninghamii var. cunninghamii	Branching Groundsel	Tdr	323
Alternanthera denticulate	Lesser Joyweed	Tda	308
Atriplex lindleyi subsp. inflata	Corky Saltbush	Tdr	302
Glinus lotoides	Hairy Carpet-weed	Tda	294
Dysphania pumilio	Clammy Goosefoot	Tdr	231
Polygonum plebeium	Small Knotweed	Tda	197
Sporobolus mitchellii	Rat-tail Couch	Tda	191
Enchylaena tomentosa var. tomentosa	Ruby Saltbush	Tdr	161
Centipeda minima subsp. minima	Spreading Sneezeweed	Atl	133
Atriplex leptocarpa	Slender-fruit Saltbush	Tdr	116
Euphorbia dallachyana	Flat Spurge	Tdr	108
Sphaeromorphaea littoralis	Spreading Nut-heads	Tda	104

In 2015–16, 10 species were recorded that had not previously been recorded at LMW wetlands in the last nine years of TLM condition monitoring (Table 4.6). Five species are from the drought-tolerant functional group (Tdr) and four species are from the amphibious monocotyledons functional group (Ate). One of the drought-tolerant species, Green Copperburr (*Sclerolaena decurrens*) is listed as vulnerable in Victoria. The amphibious species were recorded at Mulcra Lower Horseshoe and Websters Lagoon, which both held water approximately twelve months ago and are displaying a vegetation response to inundation. Three of these species have conservation significance in Victoria.

Table 4.6. Species recorded at LMW in 2015–16 that were not previously recorded over the last nine years of TLM wetland condition monitoring.

Key: CS = conservation status in Victoria: k = poorly known, v = vulnerable, FFG = listed under the Victorian *Flora and Fauna Guarantee Act 1988*; FG = functional group: Ate = amphibious monocotyledons, Tdr = drought-tolerant species.

Scientific name	Common name	CS	FG	Location recorded	Abun. (out of 2520)
Atriplex vesicaria subsp. *mactocystidia*		k	Tdr	Upper Lindsay	1
Boerhavia dominii	Tah-vine		Tdr	Websters Lagoon	1
Isolepis australiensis	Inland Club-sedge	k	Ate	Mulcra Lower Horseshoe	5
Isolepis congrua		v; FFG	Ate	Mulcra Lower Horseshoe	49
Juncus usitatus			Ate	Websters Lagoon	2
Lipocarpha microcephala	Bottom Rush	v	Ate	Mulcra Lower Horseshoe	23
Nicotiana occidentalis			Tdr	Lake Wallawalla	2
Portulaca oleracea	Common Purslane		Tdr	Lake Wallawalla	3
Sclerolaena decurrens	Green Copperburr	v	Tdr	Lake Wallawalla	1
Sclerolaena obliquicuspis	Limestone Copperburr		Tdr	Bilgoes Billabong	2

4.4.2 Non-native species

As different numbers of wetlands were surveyed in different years, non-native species were assessed as a proportion of species abundance (Figure 4-2). The greatest percentage of non-native species were recorded in 2012–13. Non-native species were less than 5% of the total proportion of abundance in 2015–16.

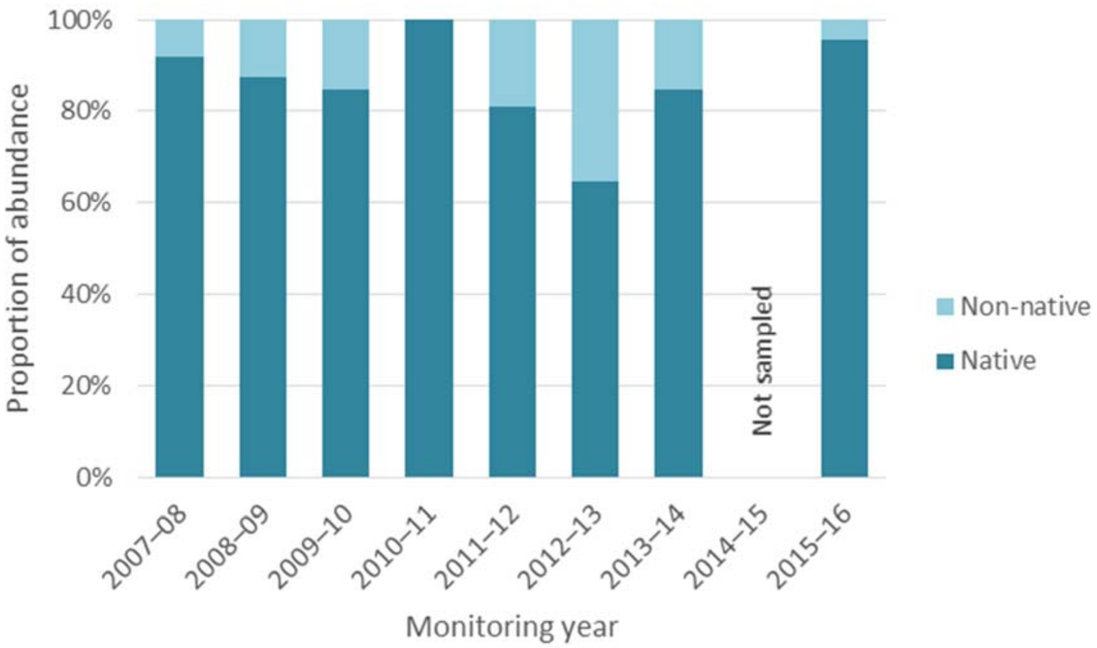


Figure 4-2. Proportion of non-native species abundance per wetland at LMW in all monitoring years (*n* = 11 in 2007–08 and 2008–09; *n* = 9 in 2010–11; *n* = 12 in all other years).

4.4.3 Rare or threatened species

In 2015–16, of the 109 species recorded, 16 are listed as having conservation significance in Victoria (Table 4.7). Four of these species had not previously been recorded at the LMW icon site during TLM wetland condition monitoring.

Surveys undertaken for TLM do not specifically target species with conservation significance. The abundance count for each species (out of a possible 2520, being the number of 1 m x 1 m cells surveyed at LMW wetlands) is included in Table 4.7. For comparative purposes, the most abundant species recorded at LMW wetlands in 2015–16 was Common Sneezeweed (*Centipeda cunninghamii*), which had an abundance count of 332. More information on rare or threatened species recorded at LMW wetlands in previous monitoring years can be found in section 3 in Part B of this report.

Table 4.7. Rare or threatened plant species recorded at LMW wetlands in 2015–16.

Key: FG = functional group: Arp = amphibious floating species, Ate = amphibious monocotyledons, Tda = terrestrial species that typically occur in damp habitats, Tdr = drought-tolerant species; CS = conservation status in Victoria, v = vulnerable, r = rare, k = poorly known, FFG = listed under the Victorian *Flora and Fauna Guarantee Act 1988*; ^Habitat preference is from Cunningham et al (1992) and Harden (1992, 1993, 2000, 2002), and notes from previous TLM records.

FG	Scientific name	Common name	Family	CS	Recorded at	Habitat preference^	Abundance (out of 2520)
Arp	*Ammania multiflora*	Jerry-jerry	Lythraceae	v	Lake Wallawalla	Grows in wet or damp conditions, often in shallow water of swamps and on river bank areas with heavy clay soils. Recorded at numerous wetlands at each of the islands in various monitoring years.	13
Tda	*Asperula gemella*	Twin-leaf Bedstraw	Rubiaceae	r	Bottom Island Lake Wallawalla	Grows inland on damp sites, chiefly along water courses in clay soils. Recorded once previously at Bottom Island (2011–12) and once at Crankhandle (2009–10).	2
Tdr	*Atriplex vesicaria* subsp. *macrocystidia*		Chenopodiaceae	k	Upper Lindsay	Grows on flat saline or gypseous plains. Recorded for the first time at TLM LMW wetland sites during 2015–16 surveys.	1
Tda	*Austrobryonia micrantha* (formally known as *Mukia micrantha*)	Mallee cucumber	Cucurbitaceae	r	Lake Wallawalla Mulcra Lower Horseshoe Wetland 33	A species that typically occurs on grey clay soils along floodplains, in areas where flood waters have receded and around lakes. Recorded at numerous wetlands in various monitoring years, particularly at Mulcra Upper Horseshoe.	12
Tdr	*Calotis cuneifolia*	Blue Burr-daisy	Asteraceae	r	Lilyponds	Grows mostly in sandy and red clay loam soils in a wide range of plant communities. Recorded in the previous monitoring year (2013–14) at Lilyponds	5

FG	Scientific name	Common name	Family	CS	Recorded at	Habitat preference^	Abundance (out of 2520)
						and on one occasion at Upper Mullaroo (2011–12).	
Ate	*Cyperus pygmaeus*		Cyperaceae	v	Mulcra Lower Horseshoe	Grows on banks and seasonally wet floodways on clay soils in open locations. Recorded once at Mulcra Upper in 2008–09, then again in 2011–12 at Bottom Island, Upper Lindsay and Mulcra Upper Horseshoe.	26
Ate	*Isolepis australiensis*	Inland Club-sedge	Cyperaceae	k	Mulcra Lower Horseshoe	Grows in seasonally wet situations. Recorded for the first time at TLM LMW wetland sites during 2015–16 surveys, though known to occur at Hattah Lakes.	5
Ate	*Isolepis congrua*		Cyperaceae	v; FFG	Mulcra Lower Horseshoe	Grows in seasonally wet situations. Recorded for the first time at TLM LMW wetland sites during 2015–16 surveys.	49
Ate	*Lipocarpha microcephala*	Bottom Rush	Cyperaceae	v	Mulcra Lower Horseshoe	Grows in damp places such as drying lake margins. Widespread but scattered and uncommon. Recorded for the first time at TLM LMW wetland sites during 2015–16 surveys.	23
Tdr	*Malacocera tricornis*	Goat Head	Chenopodiaceae	r	Lake Wallawalla	Found on clay pans and floodplains with heavy soils. Only recorded once previously, in 2009–10, also at Lake Wallawalla following environmental watering.	3
Tda	*Phyllanthus lacunarius*	Lagoon Spurge	Phyllanthaceae	v	Crankhandle	Occurs on most soil types in creek beds, on banks and on floodplains. Recorded at various locations on Lindsay Island across monitoring years.	1

FG	Scientific name	Common name	Family	CS	Recorded at	Habitat preference^	Abundance (out of 2520)
Tdr	*Sarcozona praecox*	Sarcozona	Aizoaceae	r	Lake Wallawalla	Grows in arid areas on heavy or sandy soils. Recorded once previously at Lake Wallawalla in 2013–14.	2
Tdr	*Sclerolaena decurrens*	Green Copperburr	Chenopodiaceae	v	Lake Wallawalla	Occurs on low rises on the floodplain. Recorded for the first time at TLM LMW wetland sites during 2015–16 surveys.	1
Tdr	*Sclerolaena muricata* var. *muricata*	Black Roly-poly	Chenopodiaceae	k	Bilgoes Billabong Bottom Island Crankhandle Lake Wallawalla Upper Mullaroo Websters Lagoon	Widespread colonising species in NSW, especially on overgrazed or overstocked areas on heavy soils. Recorded at numerous wetlands in most monitoring years, particularly on Lindsay Island.	87
Tdr	*Senecio cunninghamii* var. *cunninghamii*	Branching Groundsel	Asteraceae	r	Bottom Island Crankhandle Lilyponds Mulcra Lower Horseshoe Mulcra Upper Horseshoe Scotties Billabong Upper Lindsay Websters Lagoon Wetland 33	Grows in various habitats; open plains on grey clay and clay loam soils. This species is abundant at LMW, particularly on Lindsay Island.	323
Tda	*Solanum lacunarium*	Lagoon Nightshade	Solanaceae	v	Upper Mullaroo	Found on heavy clay soils on the floodplain. Found in low abundance in scattered locations at Lindsay Island across survey years.	3

4.4.4 Functional groups

Functional group abundance data are displayed for wetlands surveyed over four monitoring years (Figure 4-3). Monitoring years were chosen to display changes in community composition according to inundation events.

The first year of monitoring occurred in 2007–08 during the millennium drought, though effects from environmental water delivered in 2005 and 2006 were evident through the presence of species from amphibious functional groups (Arp, Ate, Atl). Following drawdown of the natural flood (2010–11), species from an array of functional groups were recorded in 2011–12. In 2013–14, the abundance of species from inundation-responsive functional groups (e.g. amphibious and Tda functional groups) declined in line with expectations, as wetlands continued to dry out following inundation. In 2015–16, the highest abundance was recorded for drought-tolerant species (Tdr functional group), although species from an array of functional groups were recorded, reflecting the various states of inundation across the icon site.

Functional group abundance data for each wetland in 2015–16 are displayed in Table 4.8. Abundance in a variety of functional groups was particularly high across Mulcra Island. Some species from the floating functional group were recorded at Bottom Island where water from the Lindsay River was beginning to trickle into the wetland. Long-dry wetlands were dominated by drought-tolerant species (Tdr).

Table 4.8. Functional group abundance data for each wetland at LMW in 2015–16.

Key: Inundation status: long-dry = wetland was dry at the time of the survey and had not held water for more than two years previously; 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. Functional groups: A = amphibious plants, Arp = amphibious floating plants, Ate = amphibious herbs, Atw = amphibious woody plants, F = floating plants, T = terrestrial species, Tda = terrestrial plants preferring damp habitats, Tdr = drought-tolerant plants.

Wetland	Inundation status	Functional group								
		F	Arp	A	Ate	Atl	Atw	T	Tda	Tdr
Bilgoes Billabong	Long-dry						6	1	31	293
Bottom Island	Long-dry	15	15		6	3	3	1	4	69
Crankhandle	Long-dry				13				23	97
Lake Wallawalla	Drawdown		13	4	17	31	39	3	258	212
Lilyponds	Long-dry				4		27	2	18	251
Mulcra Lower Horseshoe	Intermittent-dry			2	105	180	78	1	859	315
Mulcra Upper Horseshoe	Long-dry			14	15	258	42		229	274
Scotties Billabong^	Long-dry								5	56
Upper Lindsay	Long-dry				2		14		8	179
Upper Mullaroo	Long-dry				1				4	352
Websters Lagoon	Intermittent-dry		4		2	2	9	1	57	47
Wetland 33	Intermittent-dry					39	47		131	135

^Due to the current sampling methods abundances at Scotties Billabong are likely to be lower than at other wetlands (only four quadrats were surveyed at Scotties Billabong, compared with twelve at Bilgoes Billabong).

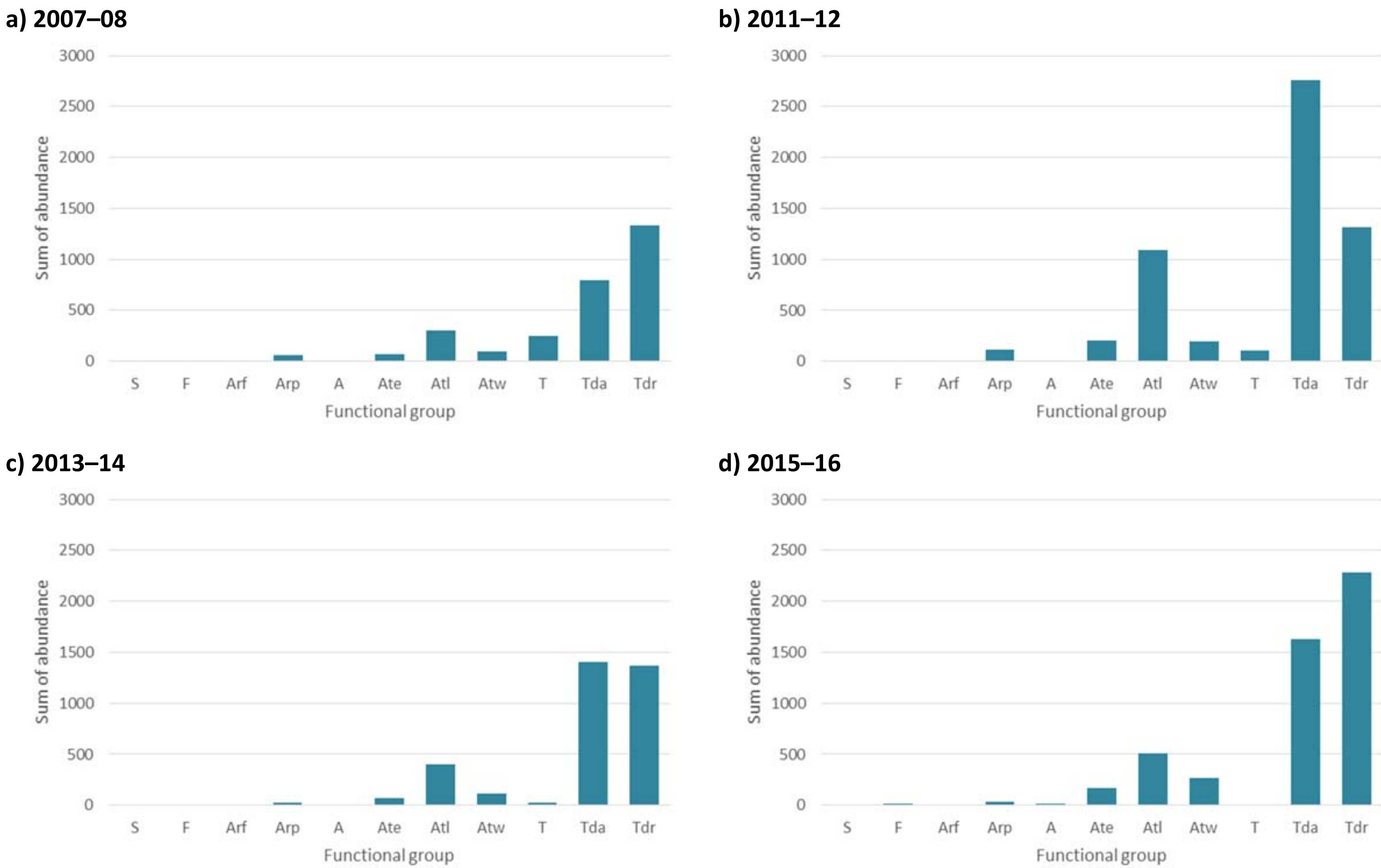


Figure 4-3. Functional group abundance data for all wetlands over four monitoring years at LMW (n = 11 in 2007–08; n = 12 in 2011–12, 2013–14 and 2015–16).

4.4.5 Community composition

Community composition differs between WRCs and monitoring years. As there was a significant interaction between year and WRC (*P* = 0.0001), PERMANOVA pairwise tests were undertaken, based on mean composition and abundance data per transect, per site, per year for each WRC (Table 4.9). The functional group communities present at ephemeral wetlands in 2015–16 were significantly different to the communities present in all other monitoring years except 2013–14.

Table 4.9. PERMANOVA (pairwise) test results showing differences in functional group composition and/or abundance between years for each WRC. Significance has been reported using Monte Carlo *P*-values and statistically significant effects are indicated in bold (α ≤ 0.05).

WRC	2015–16 vs 2007–08	2015–16 vs 2008–09	2015–16 vs 2009–10	2015–16 vs 2010–11	2015–16 vs 2011–12	2015–16 vs 2012–13	2015–16 vs 2013–14
Semi-permanent wetlands	0.3926	**0.0138**	**0.0019**	**0.0001**	0.1459	**0.0369**	0.246
Ephemeral wetlands	**0.0186**	**0.0011**	**0.0111**	**0.0001**	**0.0001**	**0.0003**	0.0549

The MDS ordination (Figure 4-4) displays changes in functional group composition over time. In 2010–11, wetlands were inundated from natural flooding and community composition was dominated by species from the floating (F) functional group. Species from the submerged (S) and floating (F, Arf and Arp) functional groups were recorded in 2008–09 and 2009–10 at wetlands that received environmental water. In all other monitoring years, species from an array of terrestrial and amphibious functional groups were recorded.

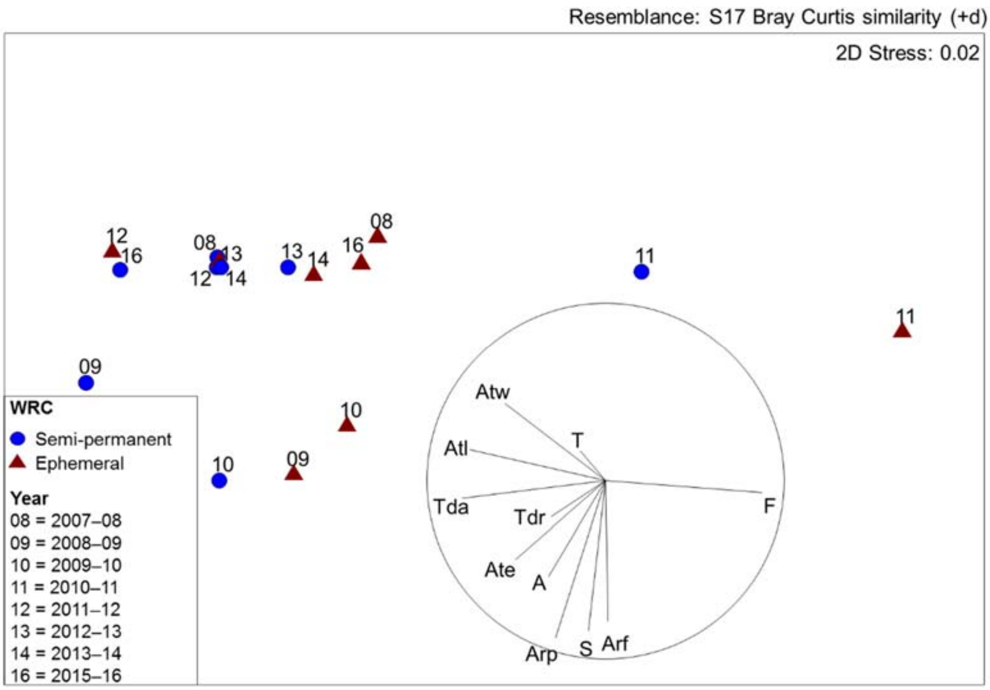


Figure 4-4. Differences in vegetation functional group composition at LMW between WRCs and monitoring years.

4.5 Discussion

4.5.1 Species richness and abundance

The majority of wetlands surveyed at LMW in 2015–16 were long-dry (i.e. eight of the twelve wetlands surveyed). Three wetlands were intermittent-dry (i.e. had held water in the last two years) and one wetland, Lake Wallawalla, was drawing down after receiving environmental water in spring 2015. These four wetlands provided damp/wet habitats favourable for wetland vegetation.

A total of 109 species were recorded at LMW in 2015–16, which was an increase from the previous monitoring year (93 species recorded in 2013–14). The most abundant species in 2015–16 was Common Sneezeweed, an amphibious herb (functional group Atl) that has emerged in response to recent inundation. Of the 109 species, 10 were not previously recorded at LMW over nine years of TLM wetland condition monitoring. Though new to this dataset, three species had been previously recorded at LMW in other studies: Bottom Rush (*Lipocarpha microcephala)* (Johns & Campbell 2011), Tah-vine (*Boerhavia dominii*) and *Juncus usitatus* (Bayes et al. 2010). After almost a decade of condition monitoring, it is interesting that new species are still being added to the TLM wetland dataset. Eight of these species were located at wetlands that either held water less than two years ago (e.g. Websters Lagoon, Mulcra Lower Horseshoe, Wetland 33) or were currently inundated (Lake Wallawalla, which was drawing down during monitoring as shown in Figure 4-5). Drying lakebeds provide a wet/dry ecotone which is particularly high in species diversity (Brock & Casanova 1997). The increase in species diversity and addition of species not previously recorded during this program are largely attributed to a response to recent inundation at four of the twelve wetlands surveyed. It is likely that the addition of new species to the TLM wetland dataset is a reflection of the variety of habitats surveyed each year (e.g. different wet/dry status at each wetland), the timing of surveys in relation to inundation (e.g. length of time since filling) and the transient nature of wetland plant communities (e.g. water-responsive species generally only live for about one year).

In 2015–16, of the 109 species recorded, 16 were listed as having conservation significance in Victoria. This is the second highest record of rare species across all monitoring years, with the highest (total of 17) recorded in 2011–12 following drawdown of the natural flood. Rare species were recorded across both semi-permanent and ephemeral wetlands and included species from a variety of functional groups (e.g. drought-tolerant and amphibious). Five rare species recorded this year were new to the TLM wetland dataset: *Atriplex vesicaria* subsp. *macrocystidia*, Inland Club-sedge (*Isolepis australiensis*), *Isolepis congrua*, Bottom Rush and Green Copperburr.

In 2015–16, non-native species were less than 5% of proportional species abundance, which is the lowest since monitoring began in 2007–08 (with the exception of 2010–11, when only six species, all native, were recorded in total due to the extent of inundation). This is attributed to the majority of wetlands at LMW being long-dry. Most non-native species were recorded at wetlands that had held water less than two years ago and few were recorded at long-dry wetlands. For wetland ecosystems in Victoria, the average relative non-native species cover is 15–20% and relative non-native species richness is ~35% (Catford & Jansson 2014). Relative non-native species richness recorded at LMW in all years was below the state average for wetlands and, with the exception of 2011–12 and 2012–13, the proportion of non-native species abundance was less than the average for Victoria.

Figure 4-5. Emerging amphibious vegetation on the edge of Lake Wallawalla, shown here drawing down from environmental water delivered in spring 2015 (F Freestone, February 2016).

4.5.2 Community composition based on functional groups

The use of functional groups is a widely accepted method of interpreting changes in plant communities in relation to disturbance, while minimising the effects of changes in species composition or inconsistencies in taxonomic classification (Brock & Casanova 1997; Campbell et al. 2014). Functional groups demonstrate the impact of flood inundation on community composition. Based on functional group data, drought-tolerant species (Tdr) were the most abundant at LMW in 2015–16. This was because 8 of the 12 wetlands surveyed had not held water within the last two years. Overall abundance was particularly high at Mulcra Island. Mulcra Lower and Mulcra Upper Horseshoes recorded high abundance of species from an array of functional groups (e.g. Tda, Tdr and Atl). These two wetlands (part of the same horseshoe) have benefited from multiple environmental flows delivered between 2005 and 2013, as well as from the natural flood in 2010–11 (Figure 4-6).

Though numerous water-responsive species were recorded at Mulcra Upper Horseshoe in response to recent environmental flows, other long-dry wetlands were strongly dominated by drought-tolerant species. Six of the thirteen most common species recorded at LMW were from the drought-tolerant functional group (Tdr). These results are in line with expectations for a drying wetland community. The drying process in arid-zone temporary wetlands is important for wetland vegetation communities as well as chemical and nutrient cycling processes (Boulton et al. 2014). However, before specific recommendations about the optimal inter-flood timing for future flows at these wetlands can be made, it is recommended that the WRCs for wetlands at LMW be reviewed. The Ecological Associates report (2007) identifies two WRCs (semi-permanent and ephemeral) at LMW, however the majority of wetlands surveyed are not specifically mentioned, or classified, in the report. Reference is made to Wetland 33 not retaining water for a long period of time, but this wetland remained inundated for approximately three years following the natural flood in 2010–11, while almost all other wetlands had drawn down. Appropriate classifications will greatly assist in future analysis and reporting. Due to the limited hydrological data available for wetlands at this icon site, it is recommended that wetlands be analysed as one WRC in future monitoring years (Brown et al. 2015).

Figure 4-6. Amphibious vegetation emerging at Mulcra Lower Horseshoe in response to environmental water delivered in 2013. This wetland has benefited from multiple environmental flows as well as the natural flood in 2010–11. The extended period of inundation appears to have killed a large proportion of River Red Gum trees, which had invaded the base of the wetland during the drought (F Freestone, February 2016).

4.5.3 Recommendations

Environmental flows appear to be supporting flow-dependent rare plants at LMW. 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 (often less than 12 months) and emerge only following an episodic event (e.g. following 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.

Due to the limited hydrological data available for wetlands at this icon site, it is recommended that wetlands be analysed as one WRC in future monitoring years (Brown et al. 2015).

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 these data were collected, they have not yet been analysed. We recommend that these data be analysed, because it could provide valuable insight into the effect of environmental water on wetland vegetation at Mulcra Island.

4.5.4 Summary

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

- Wetlands were in various wet/dry states at LMW in 2015–16 (Figure 4-7). The variety of habitats, particularly at wetlands influenced by environmental watering in recent years (2012–2015), was favourable for wetland vegetation.
- Vegetation community composition was diverse at wetlands that had held water within the last two years. Vegetation included drought-tolerant, terrestrial-damp and amphibious plant species. Long-dry wetlands were typically dominated by drought-tolerant species.
- Mulcra Island was particularly high in species richness and diversity of functional groups. Wetlands surveyed at Mulcra have benefited from environmental water on multiple occasions (2005–2013) and from the natural flood in 2010–11.
- Non-native species were particularly low in abundance in 2015–16 (less than 5% of the proportion of abundance).

Figure 4-7. Wetlands were in various wet/dry states at LMW in 2015–16. Images show: Lake Wallawalla, drawing down following environmental water delivered in spring 2015 (top left); Mulcra Lower Horseshoe, which held water less than two years ago (top right); Bilgoes Billabong, which has been dry since the natural flood in 2010–11 (bottom left); Bottom Island, which was classified as long dry for reporting purposes, though water had begun trickling in from the Lindsay River during surveys (bottom right) (F Freestone, summer 2015–16).

5 Floodplain vegetation communities

FIONA FREESTONE AND CHERIE CAMPBELL

5.1 Introduction

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

5.2 Ecological objectives

The 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 is:

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

The Living Murray program is currently undergoing refinements that involve the development of operational objectives and the identification of suitable ecological indices that link back to the vision statement and ecological objective. At the time this report was compiled, operational objectives and indices for floodplain vegetation were not yet developed. As an interim measure for this monitoring period, this chapter reports on the established vision statement and overarching ecological objective by examining changes in floodplain vegetation over time. Given the emphasis on 'floodplain communities' in the vision statement we infer the importance of floodplain vegetation in the overarching ecological objective. This will be achieved by analysing species richness, non-native species (i.e. weediness) and changes in community composition of floodplain vegetation. This information may be used to assist in developing indices, or points of reference, for future condition monitoring reporting.

5.3 Methods

This section summarises the methods used to monitor and analyse species richness, non-native species and changes in floodplain community composition over time. Annual condition monitoring was carried out at 18 sites at LMW since 2007–08 (MDFRC 2011). These sites were established to represent the various vegetation communities and watering regimes that exist on the floodplain at LMW. Water regime classes were developed to assist in classifying the floodplain into areas with common ecological characteristics and watering requirements (Ecological Associates 2007). The WRCs include vegetation communities represented by species listed in specific ecological vegetation classes (EVCs). The EVCs, WRCs and vegetation communities were assigned to each floodplain site based on GIS layers and on-ground site assessments (Table 5.1). Irregularities may exist between digitised and ground-truthed boundaries in some instances. While these classifications are useful on a broad scale, it is important to remember that specific floodplain monitoring sites may not fit neatly into these classifications. Discrepancies were addressed on a case-by-case basis.

Each year, surveys were conducted between January and March following the methods described in The Living Murray: Condition Monitoring Program design for Chowilla Floodplain and the Lindsay, Mulcra and Wallpolla Islands (MDFRC 2011). Possible limitations associated with the use of these methods are described in section 4.3.

Table 5.1. The WRCs related to floodplain vegetation communities at LMW (Ecological Associates 2007; Scholz et al. 2007a).

Key: RRG = River Red Gum, BB = Black Box.

Sites	EVC	WRC	Vegetation community	Characteristics of classes
M1A M2A	EVC 106: Grassy riverine forest	Red Gum forest	RRG	Found only in areas subject to the most frequent flooding regimes. This water regime class is (historically) subject to inundation in nearly all years and is characterised by a closed canopy of tall River Red Gum.
L1A L2A L2B W1A W2A	EVC 813: Intermittent swampy woodland	Red Gum woodland	RRG	Occurs mainly in floodplain areas immediately surrounding wetlands and along water courses that are (historically) inundated by peaks in river flow during most years.
L1B L1C L2C M1B M1C M2B M2C W1B W1C W2B W2C	EVC 103: Riverine chenopod woodland EVC 823: Lignum swampy woodland	Black Box woodland	BB	Occurs in the least-frequently inundated areas of the floodplain. This WRC is dominated by Black Box with a diverse shrubby understorey of chenopods or Lignum.

5.3.1 Floodplain inundation state

The historic and current inundation state of each floodplain site provided context for data analysis. Ten of the eighteen floodplain sites were inundated during the natural flood in 2010–11 (Table 5.2). All floodplain sites are now considered long-dry (i.e. they have not been inundated by overbank flows for at least two monitoring years). Eight of the eleven Black Box understorey sites have not been inundated for approximately 20+ years.

Table 5.2. Vegetation community and hydrological state of each floodplain site in 2015–16.

Key: BB = Black Box, RRG = River Red Gum.

Vegetation community	Site	Hydrological state 2015–16	Last inundated
BB	L1B	Long-dry	~1993–94
BB	L1C	Long-dry	~1993–94
BB	L2C	Long-dry	~1993–94
BB	M1B	Long-dry	2010–11
BB	M1C	Long-dry	~1993–94
BB	M2B	Long-dry	2010–11
BB	M2C	Long-dry	~1993–94
BB	W1B	Long-dry	2010–11
BB	W1C	Long-dry	~1993–94
BB	W2B	Long-dry	~1993–94
BB	W2C	Long-dry	~1993–94
RRG	L1A	Long-dry	2010–11
RRG	L2A	Long-dry	2010–11
RRG	L2B	Long-dry	2010–11
RRG	M1A	Long-dry	2010–11
RRG	M2A	Long-dry	2010–11
RRG	W1A	Long-dry	2010–11
RRG	W2A	Long-dry	2010–11

5.3.2 Plant species identification

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

5.3.3 Data analysis

Data analysis was as described in section 4.3.3 except that, for PERMANOVA analysis monitoring year and vegetation community were used as the fixed factors. An MDS ordination plot was used to display overall differences in functional group abundance and composition between both vegetation communities and monitoring years. For display purposes the MDS ordination is based on functional group abundance data averaged for each year within each vegetation community (River Red Gum = 7 understorey sites and Black Box = 11 understorey sites).

5.4 Results

5.4.1 Species richness

A total of 59 plant species were recorded in 2015–16 across all floodplain sites at LMW (Table 5.3). The highest species richness was recorded in 2011–12 in response to widespread flooding and above-average rainfall, which occurred in the summer of 2010–11. For a comprehensive list of all plants recorded, refer to section 4 in Part B of this report.

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

Monitoring year	**2007–08**	**2008–09**	**2009–10**	**2010–11**	**2011–12**	**2012–13**	**2013–14**	**2015–16**
Number of sites surveyed	18	18	18	17	18	18	18	18
Total species recorded	44	73	86	59	107	71	78	59

The most common floodplain species recorded at LMW in 2015–16 are listed in Table 5.4. Common species are defined as those where the abundance total was greater than 100 (out of a possible 1140, being the number of 1 m x 1 m cells surveyed across the floodplain).

Table 5.4. The most common plant species recorded on the LMW floodplain in 2015–16. Tdr = drought-tolerant species.

Scientific name	**Common name**	**Functional group**	**Abundance (out of 1140)**
Atriplex lindleyi subsp. *inflata*	Corky Saltbush	Tdr	125
Disphyma crassifolium subsp. *clavellatum*	Rounded Noon-flower	Tdr	132
Enchylaena tomentosa var. *tomentosa*	Ruby Saltbush	Tdr	398
Euphorbia dallachyana	Flat Spurge	Tdr	100
Rhagodia spinescens	Hedge Saltbush	Tdr	100
Sclerochlamys brachyptera	Short-wing Saltbush	Tdr	112

Two-spined Copperburr (*Sclerolaena uniflora*), listed as rare in Victoria, was recorded for the first time in 2015–16 over nine years of TLM floodplain condition monitoring at LMW. This species was recorded on Wallpolla Island (sites W2B and W2C) in Black Box woodland.

Species richness was assessed for both River Red Gum and Black Box understorey communities. A total of 35 plant species were recorded in River Red Gum understorey sites and 37 plant species were recorded in Black Box understorey sites in 2015–16 (Figure 5-1).

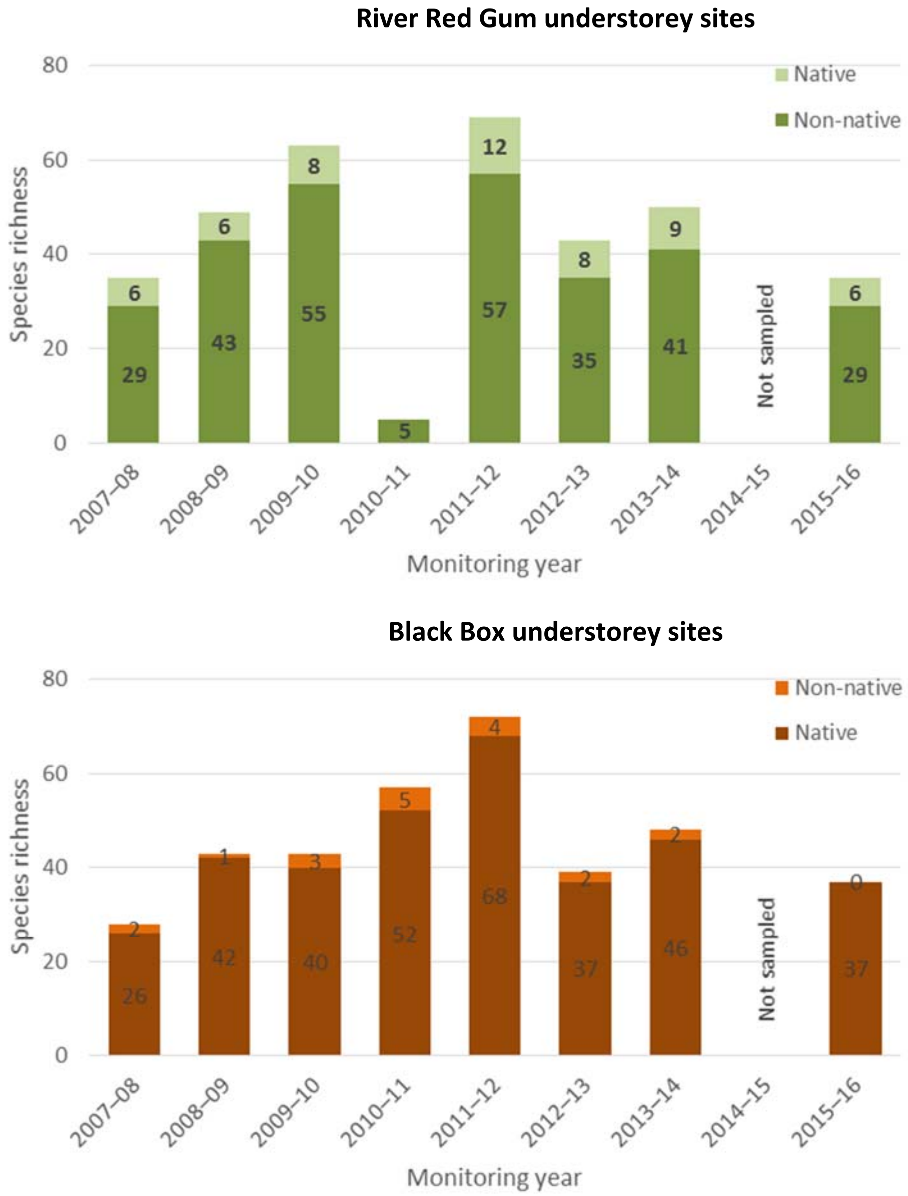
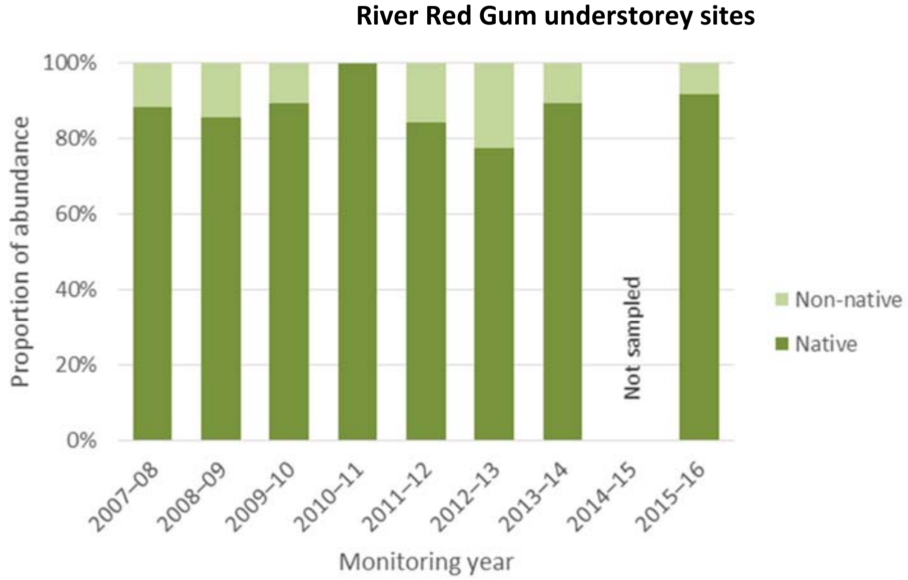


Figure 5-1. Number of native and non-native plant species recorded in River Red Gum and Black Box understorey sites at LMW in all monitoring years. Not all sites were surveyed in 2010–11 due to flooding and access issues (River Red Gum: n = 7 in all years except 2010–11, where n = 6; Black Box: n = 11 in all years).

5.4.2 Non-native species

As different numbers of sites were surveyed in different years, non-native species were assessed as a proportion of species abundance (Figure 5-2). Non-native species were less than 25% in River Red Gum communities and less than 5% in Black Box communities in every monitoring year.



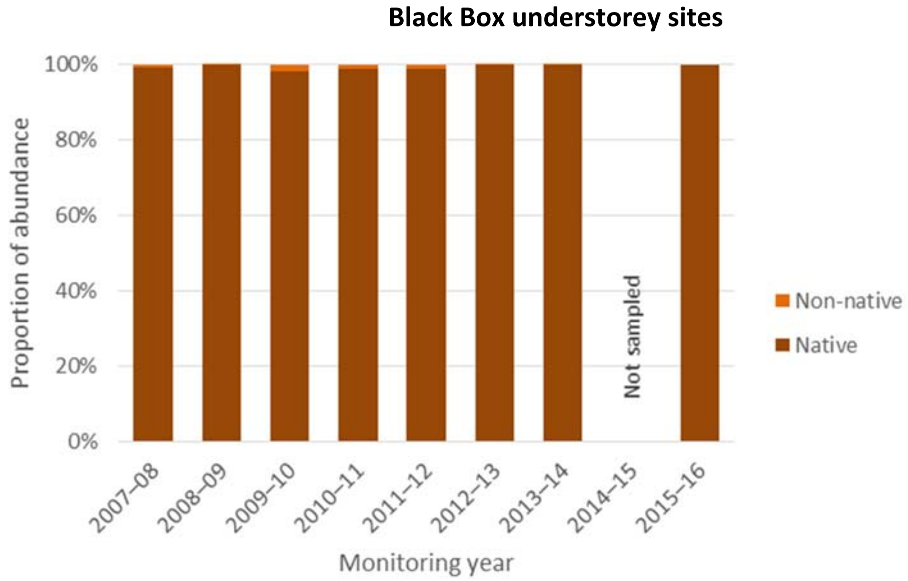


Figure 5-2. Proportion of non-native species abundance in River Red Gum and Black Box understorey communities at LMW in all monitoring years (River Red Gum: n = 7 in all years except 2010–11, where n = 6; Black Box: n = 11 in all years).

5.4.3 Rare or threatened plant species

Of the 59 species recorded in 2015–16, seven are listed as having conservation significance in Victoria (Table 5.5). Surveys undertaken for TLM do not specifically target species with conservation significance. The abundance count (out of a possible 1140, being the number of 1 m x 1 m cells surveyed on the LMW floodplain) for each species is included in Table 5.5. For comparative purposes, the most abundant species recorded on the LMW floodplain in 2015–16 was Ruby Saltbush (*Enchylaena tomentosa* var. *tomentosa*) which had an abundance count of 398. More information on rare or threatened species recorded at floodplain sites in previous monitoring years can be found in section 4 in Part B of this report.

Table 5.5. Rare or threatened plant species recorded at LMW floodplain sites in 2015–16.

Key: FG = functional group: Tda = terrestrial species that typically occur in damp habitats, Tdr = terrestrial species that typically occur in dry habitats; CS = conservation status in Victoria: FFG = listed under the Victoria *Flora and Fauna Guarantee Act 1988*, k = poorly known, r = rare, v = vulnerable; WRC = water regime class: RGF = Red Gum forest, RGW = Red Gum woodland, BBW = Black Box woodland. ^Habitat preference is from Cunningham et al (1992) and Harden (1992, 1993, 2000, 2002), and notes from previous TLM records.

FG	Scientific name	Common name	Family	CS	WRC	Habitat preference^	Abundance (out of 1140)
Tda	*Asperula gemella*	Twin-leaf Bedstraw	Rubiaceae	r	RGF RGW	Grows inland on damp sites, chiefly along water courses in clay soils. This species has been recorded in all monitoring years except 2010–11.	12
Tdr	*Calotis cuneifolia*	Blue Burr-daisy	Asteraceae	r	RGF RGW	Grows mostly in sandy and red clay loam soils in a wide range of plant communities. This species has been recorded in all monitoring years except 2010–11.	5
Tda	*Eremophila bignoniiflora*	Bignonia Emu-bush	Scrophulariaceae	v; FFG	BBW	Grows in periodically flooded heavy clay soils of river and creek floodplains, in drainage lines and near lakes. This species has been recorded in four of the eight monitoring years.	2
Tdr	*Eremophila divaricata* subsp. *divaricata*	Spreading Emu-bush	Scrophulariaceae	r	BBW	Can grow in either River Red Gum or Black Box communities on heavy clay soils of floodplains. This species is a small-to-medium woody shrub that has been recorded in every monitoring year.	20
Tdr	*Sclerolaena muricata* var. *muricata*	Black Roly-poly	Chenopodiaceae	k	BBW RGW	Widespread colonising species in New South Wales, especially on overgrazed or overstocked areas on heavy soils. This species has been recorded in every monitoring year since 2008–09.	14

FG	Scientific name	Common name	Family	CS	WRC	Habitat preference^	Abundance (out of 1140)
Tdr	*Sclerolaena uniflora*	Two-spined Copperburr	Chenopodiaceae	r	BBW	Occurs on loamy, subsaline soils. This species is very similar to *S. diacantha* and may intergrade in some areas. *S. uniflora* was recorded for the first time in 2015–16 on the LMW floodplain during TLM surveys.	9
Tdr	*Senecio cunninghamii* var. *cunninghamii*	Branching Groundsel	Asteraceae	r	BBW RGF RGW	Grows in various habitats, open plains on grey clay and clay loam soils. Widespread in inland New South Wales. This species is abundant at LMW, particularly on Lindsay Island and has been recorded in all monitoring years.	89

5.4.4 Functional groups

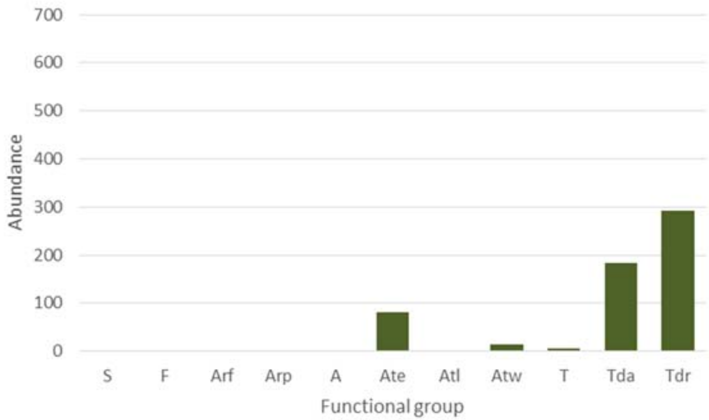
Functional group abundance data have been presented for four monitoring years for both River Red Gum (Figure 5-3) and Black Box (Figure 5-4) understorey communities. Monitoring years were chosen to display changes in community composition according to inundation disturbance.

The first year of monitoring occurred in 2007–08 during the millennium drought. The presence of the Ate functional group is strongly driven by the abundance of Tangled Lignum (*Duma florulenta*) on the floodplain (in 2007–08, Tangled Lignum was the only Ate species). In 2010–11, the natural flood inundated all of the River Red Gum and three of the Black Box understorey sites. The greatest abundance (and number of species) was recorded in 2011–12 after flood water had receded from the floodplain. The River Red Gum understorey community was dominated by species from the floating functional group in 2010–11, as the sites were inundated at the time of monitoring. In 2013–14, the presence of species in multiple amphibious functional groups indicated a continued, small response to the natural flood in River Red Gum communities. However in 2015–16, community composition was returning to one dominated by drought-tolerant species, as the floodplain remained dry.

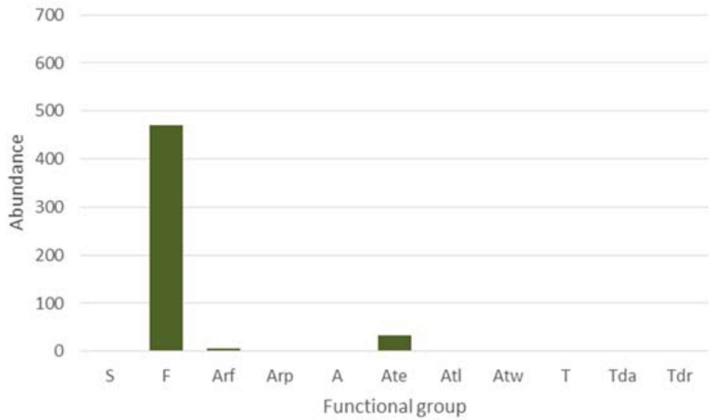
There has been very little change in Black Box understorey community composition over time. This community was heavily dominated by drought-tolerant species (Tdr functional group) in all monitoring years (Figure 5-4).

River Red Gum understorey sites

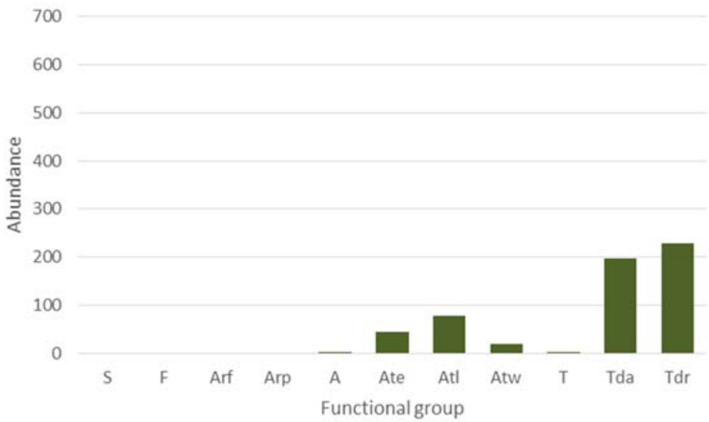
a) 2007–08



b) 2010–11



c) 2013–14



d) 2015–16

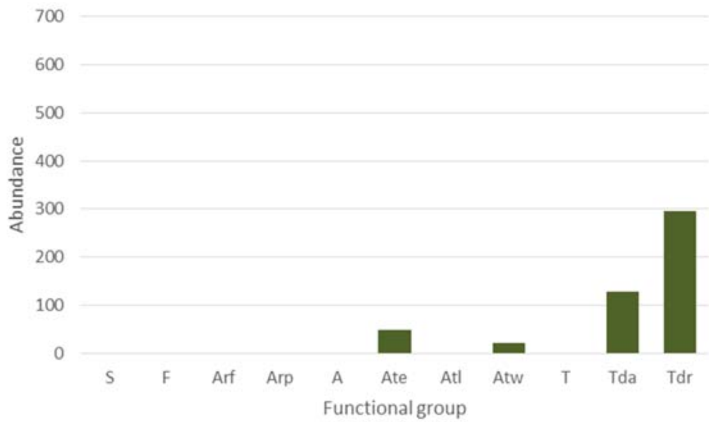
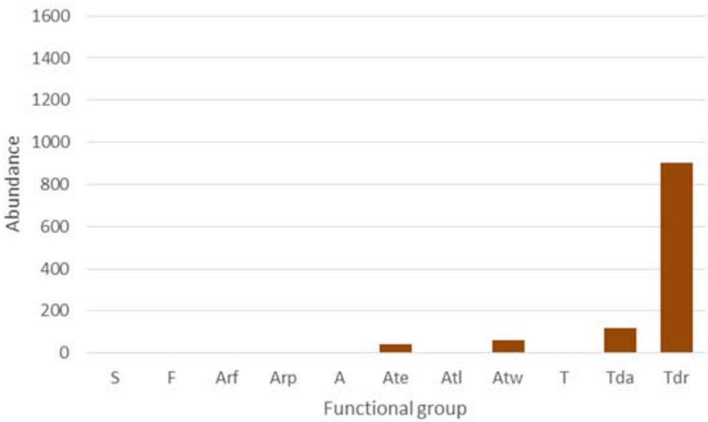


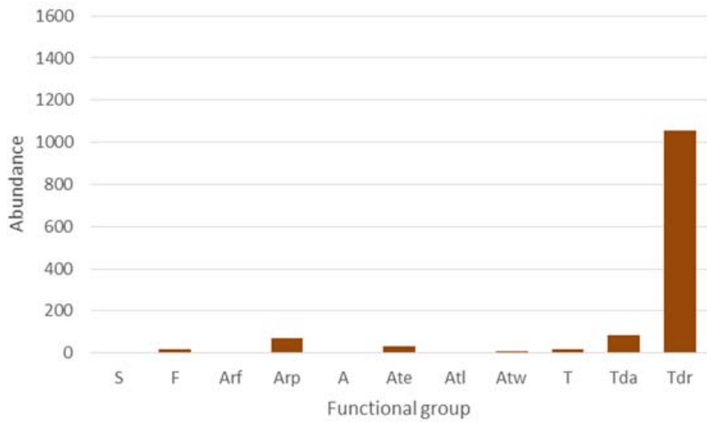
Figure 5-3. Functional group abundance data for River Red Gum understorey sites over four monitoring years at LMW (*n* = 7 sites in 2007–08, 2013–14 and 2015–16; *n* = 6 sites in 2010–11).

Black Box understorey sites

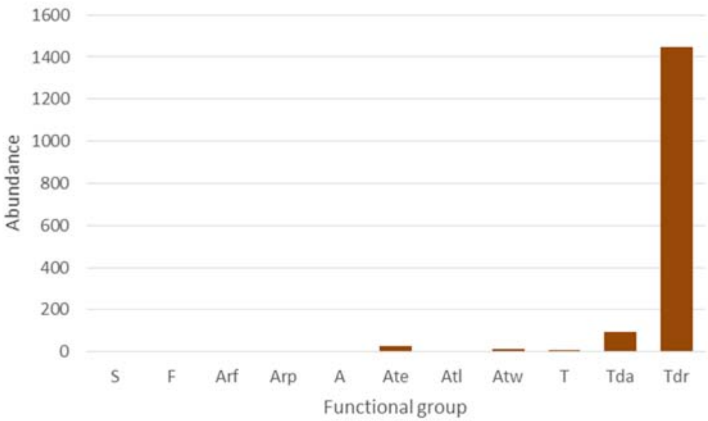
a) 2007–08

b) 2010–11





c) 2013–14



d) 2015–16

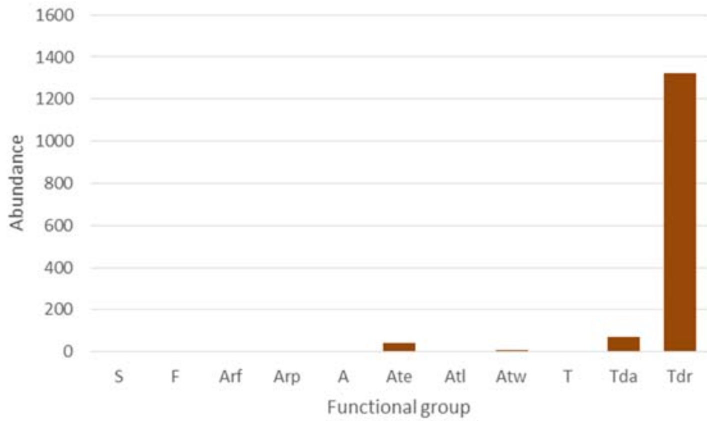


Figure 5-4. Functional group abundance data for Black Box understorey sites over four monitoring years at LMW (*n* = 11 sites).

5.4.5 Community composition based on functional groups

Community composition differs between vegetation community type and monitoring years. As there was a significant interaction ($P = 0.0001$) between year and vegetation community type (i.e. River Red Gum or Black Box understorey), PERMANOVA pairwise tests were undertaken, based on mean functional group composition and abundance data per site, per year, for each vegetation community (Table 5.6). In both understorey communities, a statistically significant difference was recorded in some, but not all monitoring years. In 2015–16, River Red Gum understorey communities were becoming more similar to that recorded in drought years (e.g. 2007–08 and 2008–09). Black Box understorey communities have changed little since 2012–13.

Table 5.6. PERMANOVA (pairwise) test results showing differences in functional group composition and/or abundance between years, for each understorey community. Significance has been reported using Monte Carlo *P*-values and statistically significant effects are indicated in bold ($\alpha \leq 0.05$).

Communities	2015–16 vs 2007–08	2015–16 vs 2008–09	2015–16 vs 2009–10	2015–16 vs 2010–11	2015–16 vs 2011–12	2015–16 vs 2012–13	2015–16 vs 2013–14
River Red Gum understorey	0.3944	0.1732	**0.0201**	**0.0001**	**0.0001**	**0.0001**	**0.0003**
Black Box understorey	**0.0018**	0.4447	0.7316	**0.0045**	**0.0031**	0.2026	0.2344

The MDS ordination in Figure 5-5a displays changes in functional group composition over time for both River Red Gum and Black Box understorey communities. In 2010–11, River Red Gum understorey sites that were inundated during the natural flood were dominated by species from the floating (F) functional group. Figure 5-5b shows a close-up of the remaining data points. River Red Gum communities in 2015–16 and during drought years (2007–08 and 2008–09), and Black Box communities in all years, were characterised by species from the terrestrial dry (Tdr) functional group.

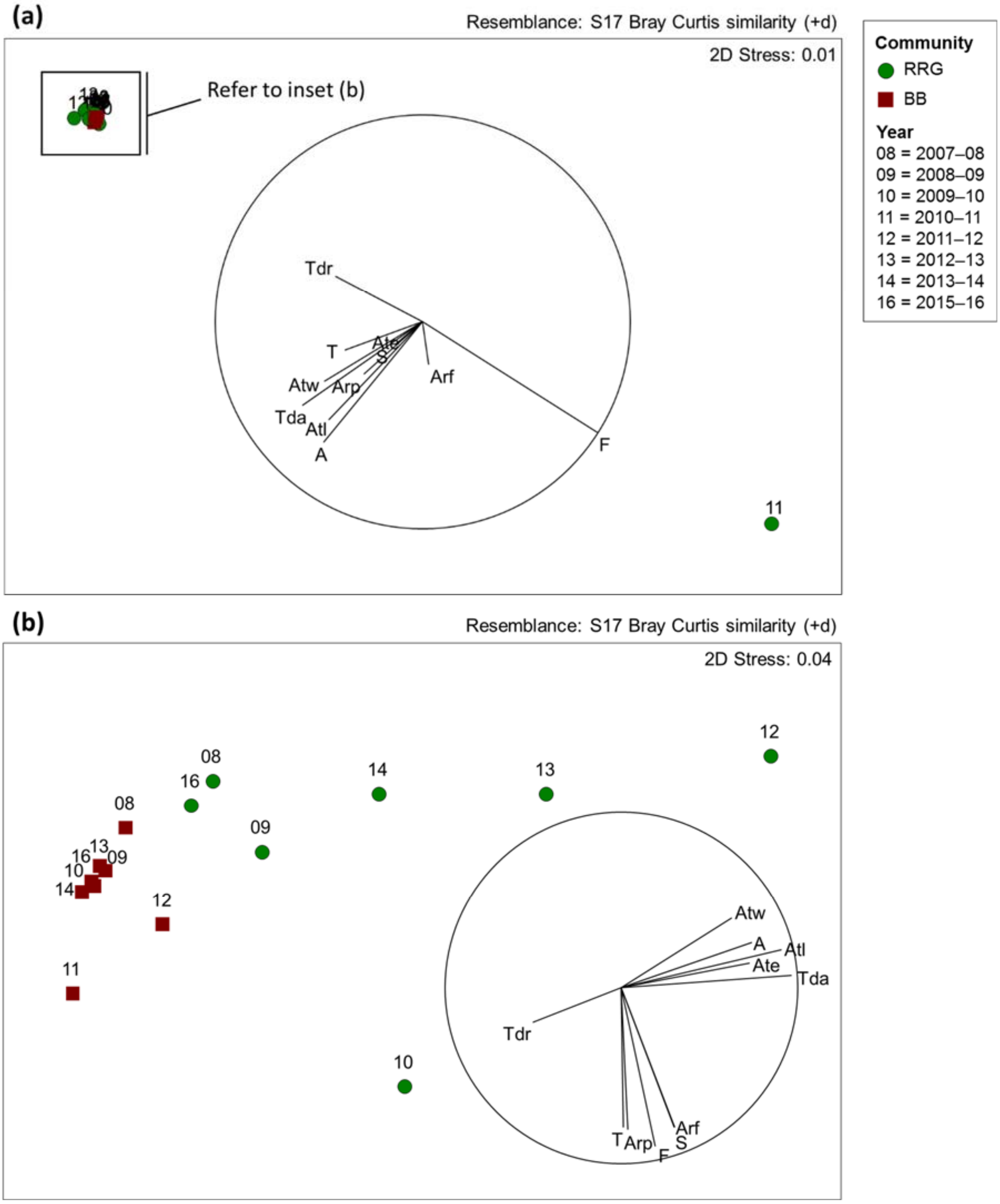


Figure 5-5. (a) Differences in vegetation functional group composition at LMW between vegetation communities and monitoring years; (b) Inset of (a). Key: RRG = River Red Gum, BB = Black Box.

5.5 Discussion

5.5.1 Species richness and abundance

The condition of understorey communities at LMW in 2015–16 is that of a dry floodplain. All sites are considered long-dry (e.g. not inundated by overbank flows for more than two years) and were heavily dominated by drought-tolerant species in both River Red Gum and Black Box understorey communities. A total of 59 plant species were recorded. Species richness has declined in each monitoring year since 2011–12, when a species richness 'boom' was recorded following the natural flood in 2010–11. The six most common species recorded in 2015–16 were drought-tolerant species (e.g. Ruby Saltbush, Rounded Noon-flower (*Disphyma crassifolium* subsp. *clavellatum*), Corky Saltbush (*Atriplex lindleyi* subsp. *lindleyi*) etc.).

Seven species recorded have conservation significance; two terrestrial species that grow in damp soils and five drought-tolerant species. It is likely that the terrestrial species that grow in damp soils (or have a preference for floodplain habitats) are being maintained through rainfall. One of the rare drought-tolerant species, Two-spined Copperburr (*Sclerolaena uniflora*), was recorded for the first time in 2015–16 over nine years of TLM floodplain monitoring at LMW. This species is listed as rare in Victoria and was found on Wallpolla Island.

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

5.5.2 Community composition

The use of functional groups is a widely accepted method of interpreting changes in plant communities in relation to disturbance, while minimising the effects of changes in species composition or inconsistencies in taxonomic classification (Brock & Casanova 1997; Campbell et al. 2014). Functional groups demonstrate the impact of flood inundation on community composition. In drought years, the floodplains at LMW were dominated by species from the terrestrial dry (Tdr) functional group. During the natural flood (2010–11), all River Red Gum sites were inundated and community composition was heavily influenced by species from the floating (F) functional group. In the following year, species from an array of terrestrial and amphibious functional groups were recorded as flood water receded from the floodplain. Following this species richness and diversity boom and as the floodplain continued to dry, River Red Gum understorey sites trended back towards a drought-tolerant community. The abundance of functional group Ate, largely driven by Tangled Lignum, has declined since monitoring began in 2007–08. However, in 2015–16, River Red Gum understorey communities were not significantly different from that recorded in drought years (2007–08 and 2008–09).

There has been little change in the community composition of Black Box understorey sites across all monitoring years. While there was an increase in species richness and abundance in the monitoring year following the natural flood (2011–12), this was largely driven by species from the terrestrial dry (Tdr) functional group. This response is likely associated with above-average rainfall that was experienced across the region the previous year (325 mm recorded over summer 2010–11 compared with a long-term summer average of 60 mm at Werrimull) (BOM 2014). Black Box understorey communities remain heavily dominated by drought-tolerant species.

5.5.3 Recommendations

The 2015–16 survey results are consistent with expectations of a drying floodplain (Rogers & Ralph 2011). 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 for as chemical and nutrient cycling processes (Boulton et al. 2014). However, intermittent inundation is required to sustain the aquatic and amphibious floodplain communities. The ideal flooding frequency for River Red Gum communities is inundation once in every three to five years (Rogers & Ralph 2011). River Red Gum communities at LMW were last inundated by overbank flows in the 2010–11 flood (five years ago). The ideal flooding frequency for Black Box woodland communities is inundation once in every 10 years (Rogers & Ralph 2011). Only three Black Box understorey sites were inundated by flood water during 2010–11. The majority of Black Box understorey sites at LMW have not been inundated for ~20 years. Both River Red Gum and Black Box floodplain understorey communities at LMW would benefit from inundation in the near future.

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 on the lower floodplain, three mid-floodplain and three high on the floodplain) were established and surveyed in 2011 (prior to delivery of environmental water) and 2013 (following delivery of environmental water). 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 these 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.

5.5.4 Summary

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

- Two-spined Copperburr, was recorded for the first time in 2015–16 over nine years of TLM floodplain monitoring at LMW. This species is listed as rare in Victoria and was found on Wallpolla Island.
- The abundance of non-native species is consistently low in all monitoring years at LMW. In particular, non-native species were less than 5% of the proportion of abundance in Black Box understorey communities.
- The condition of understorey communities at LMW in 2015–16 was that of a dry floodplain. While the drying process is important for understorey vegetation communities in arid floodplains, intermittent inundation is required to sustain aquatic and amphibious plant species. River Red Gum and Black Box understorey communities would benefit from inundation in the near future (Figure 5-6). Proposed infrastructure at this icon site may benefit some of these communities.

Figure 5-6. The floodplain at LMW in 2015–16 was largely dominated by drought-tolerant species such as Hedge Saltbush (*Rhagodia spinescens*) and Ruby Saltbush (*Enchylaena tomentosa* var. *tomentosa*) (F Freestone, January 2016).

6 Lignum

FIONA FREESTONE

6.1 Introduction

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

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

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

6.3 Methods

Lignum condition was assessed using two methods;

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

6.3.1 Sites

There are 10 established Lignum monitoring sites on each island at LMW (Lindsay: L1 to L10, Mulcra: M1 to M10, Wallpolla: W1 to W10), each containing 30 tagged Lignum plants. Sites were selected to represent the various EVCs, WRCs and elevations of the floodplain. Sites L1 to L5, M1 to M5 and W1 to W5 were established in 2006–07 and were reassessed in summer 2007–08 and 2008–09 and in spring annually for subsequent years (excluding 2014–15 when no data were collected due to changes in funding). An additional 16 sites were established in 2012–13 to provide a more representative sample of the LMW icon site (L6 to L10, M6 to M10, W6 to W10, L3A, L5A, M1A and W3A). To compensate for the loss of plants through mortality (due to extended drought) or missing plants (e.g. plants that were not able to be located due to missing tags and recorded as no data), an additional 143 plants were added to the survey in 2012–13. These plants were located as geographically close to the original plants as possible and were monitored in 2012–13, 2013–14 and 2015–16 and will continue to be surveyed annually in spring. Location details, maps and photos taken at each site in each survey year are provided in section 5 of Part B of this report.

6.3.2 Hydrology

Site hydrology was determined in order to provide context to condition monitoring data. Flooding history for each site was estimated using RiM-FIM (Overton et al. 2006) and Murray River flows recorded upstream of Lock 9 (data courtesy of MDBA). Data recorded upstream of Lock 9 were deemed sufficient for all sites at LMW for the purposes of this report, as diversions in a major flood are unlikely to substantially impact the extent of flood water in relation to Lignum sites.

Beginning in spring 2010, natural flooding of the Murray River and LMW Islands and its associated floodplains resulted in the inundation of 18 Lignum sites (Table 6.1). Based on RiM-FIM and digital elevation modelling, the remaining sites were most likely last inundated in either 1993 or 1975 and have received only rainfall since. As there are spatial limitations associated with using RiM-FIM, in some instances site observations were used where inaccuracies were evident in GIS data.

Site W1, near Wallpolla Horseshoe, is the only site at LMW Islands to have received environmental watering, which occurred in 2005, 2006, 2008, 2009 and 2015. In addition, this site was inundated for an extended period between 2008 and 2013. Though it was dry during monitoring in 2013–14, during the 2015–16 surveys this site was again inundated to approximately 30 cm depth.

A regulating structure was constructed on lower Potterwalkagee Creek at Mulcra Island in 2010, 100 m downstream of site M1. This structure retained and elevated flood water in 2011 and 2012, submerging Lignum plants at this site. Similarly, in 2013–14, plants were submerged again when environmental flows were retained by the regulator.

Table 6.1. Time since last natural flood for each lignum site at LMW Islands. Ticks indicate the most recent natural flood to inundate each site.

Site	Last natural flood		
	1975	1993	2011
L1		✓^	
L2			✓*
L3		✓^	
L3A		✓	
L4		✓	
L5		✓	
L5A		✓	
L6			✓
L7		✓	
L8			✓*
L9			✓*
L10			✓

Site	Last natural flood		
	1975	1993	2011
M1			✓
M1A			✓
M2			✓
M3		✓	
M4		✓	
M5		✓	
M6			✓
M7			✓
M8			✓*
M9		✓	
M10			✓*

Site	Last natural flood		
	1975	1993	2011
W1			✓
W2			✓*
W3	✓		
W3A			✓
W4		✓	
W5		✓	
W6			✓*
W7			✓*
W8			✓*
W9	✓		
W10	✓#		

*Observations confirm this site was inundated in 2011 although RiM-FIM data differs.
^Observations confirm this site did not flood in 2011 although RiM-FIM data differs.
#According to RiM-FIM, this site was last flooded in 1975, however observations confirm pooling of water in 2011, possibly from excessive rainfall.

6.3.3 Analysis

The Lignum Condition Index (LCI) method used for assessing the condition of Lignum at LMW Islands is outlined in section 7 of The Living Murray: Condition Monitoring Program design for Chowilla Floodplain and the Lindsay, Mulcra and Wallpolla Islands (MDFRC 2011). The LCI is comprised of two plant condition indicators: viability and colour (Table 6.2). In some survey years, plants were not able to be located, either due to missing tags or inundation. Plants that were not able to be located and assessed were recorded as no data. During 2015–16 surveys, only 10 of the 30 previously tagged plants at W3A were able to be located and surveyed as the site was disturbed by cattle on Wallpolla Island.

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

% viable	score
> 95	6
75 ≤ 95	5
50 ≤ 75	4
25 ≤ 50	3
5 ≤ 25	2
0 ≤ 5	1
0	0

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

Condition over time

To evaluate change in Lignum condition over time, scores for each of the two LCI indicators for each plant were added together and then categorised into one of five condition categories (Table 6.3). Change in lignum condition over 10 years was assessed using data from sites established in 2006–07 (L1 to L5, M1 to M5 and W1 to W5). One-way analysis of variance (ANOVA) was used to determine if there were statistically significant differences in mean LCI per year. Holm-Sidak pairwise comparisons (SigmaPlot version 11; Systat Software, San Jose, CA) were then undertaken to determine which pairs of years were significantly different.

Table 6.3. Condition categories for LCI combined scores.

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

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

Three-survey comparison

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

6.4 Results

6.4.1 *Condition over time*

Condition results for plants that were assessed annually from 2006–07 to 2015–16 at sites L1 to L5, M1 to M5 and W1 to W5 (*n* = 450) are presented in Figure 6-1. There has been an annual increase in the number of plants with an LCI score of zero (excluding data from 2010–11, which contains a large proportion of no-data values associated with inundation and inaccessible sites), with the highest incidence recorded for 2015–16. In 2015-16, approximately 56% of surveyed plants were recorded as zero (dead or dormant). The majority of plants that had viable above-ground biomass were recorded as being in moderate or good condition in 2015–16.

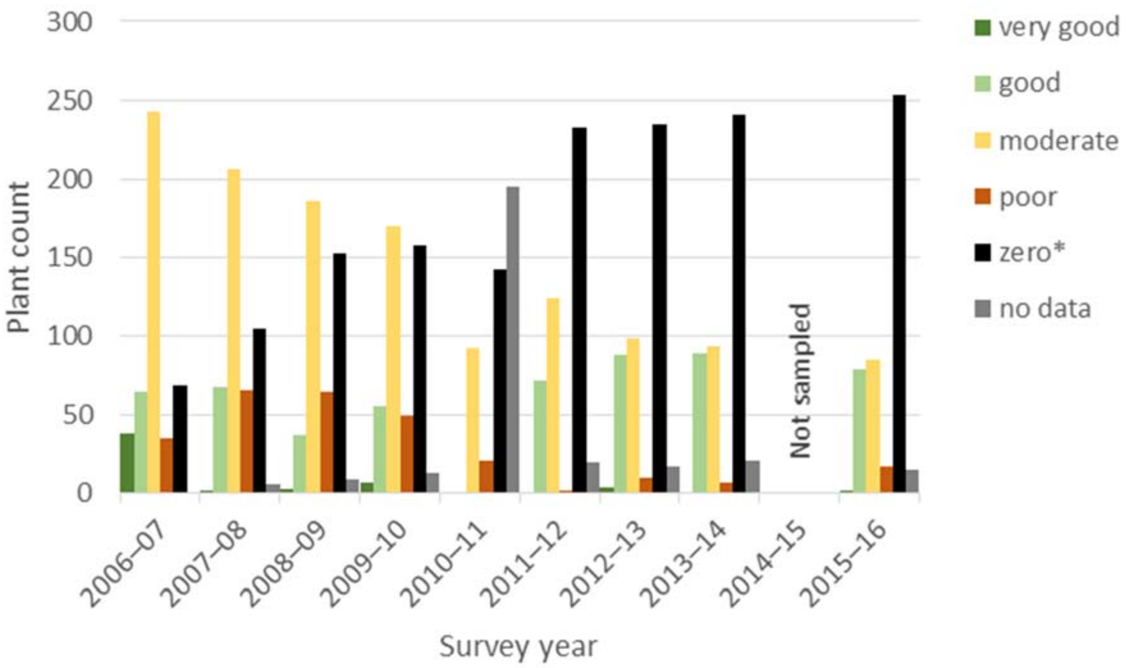


Figure 6-1. Count of plants in each LCI condition category across the LMW icon site for each survey year (sites L1 to L5, M1 to M5 and W1 to W5; n = 450 plants per year).

Key: *the category zero relates to Lignum plants that were observed to have no viable above-ground biomass. These plants were presumed to be either dead, or dormant and persisting as viable underground rootstock. Note: in 2012–13 there were 235 Lignum plants recorded as zero for two or more years and showed no sign of regeneration following the 2010–11 flood. As these plants showed no sign of recovery following favourable environmental conditions, they were classified as dead and were not specifically assessed in 2013–14 and 2015–16. For the purposes of this graph, these 235 plants are included in the zero category data in 2013–14 and 2015–16, even though they were not explicitly surveyed.

The LCI scores were pooled per survey to determine average Lignum condition for each year. Figure 6-2 shows the change in average Lignum condition per year from 2006–07 to 2015–16 (excluding 2014–15). Condition has been maintained at a similar level since 2011–12.

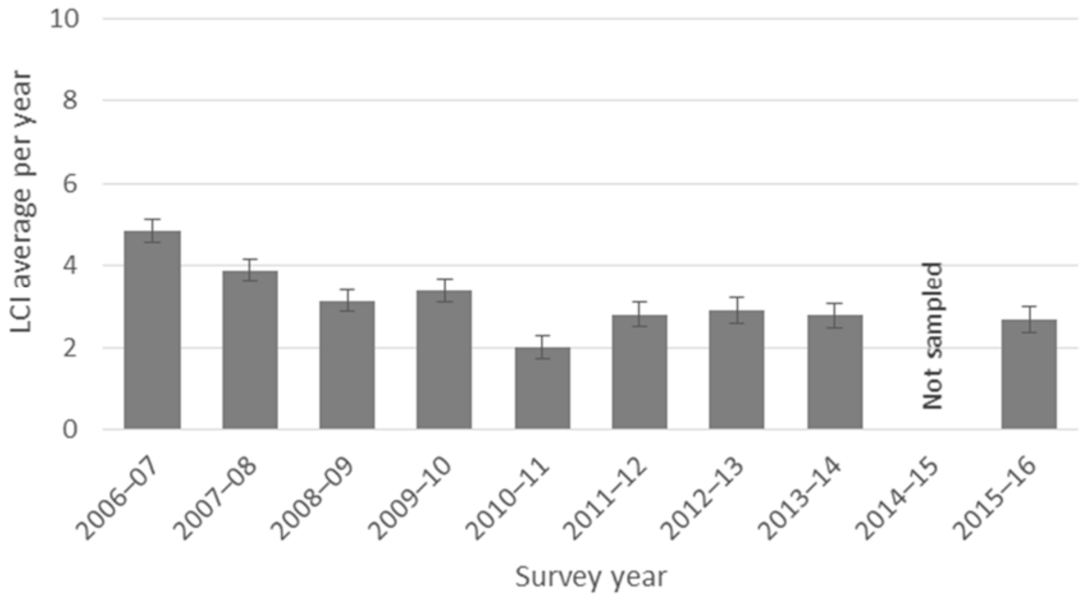


Figure 6-2. Average (± 95% CI) LCI for each survey at the LMW icon site (sites L1 to L5, M1 to M5 and W1 to W5; n = 450 plants per year).

One-way ANOVA using mean LCI scores identified a significant effect of survey years ($P < 0.001$). Holm-Sidak pairwise comparisons showed statistically significant differences between some, but not all years (at the $P < 0.05$ level). Pairwise comparisons for the current survey year (2015–16) and all other survey years are presented in Table 6.4. There was no statistically significant difference between 2015–16 and each of the previous four surveys.

Table 6.4. Holm-Sidak pairwise comparisons *P*-values, showing statistically significant differences (**bold**) between 2015–16 and all other survey years at LMW (L1 to L5; M1 to M5; W1 to W5).

Survey year	2006–07	2007–08	2008–09	2009–10	2010–11	2011–12	2012–13	2013–14
2015–16	**<0.001**	**<0.001**	0.212	**0.007**	0.089	0.978	0.865	0.933

6.4.2 *Three-survey comparison*

The condition of plants at all sites (L1 to L10, M1 to M10, W1 to W10, including all replacement sites and plants and excluding dead or missing plants that were discontinued in 2012–13) were compared between survey years 2012–13, 2013–14 and 2015–16 (*n* = 910) (Figure 6-3). Between 2013–14 and 2015–16, there was a decline in the number of plants recorded in good condition and an increase in the number of plants recorded in moderate condition. In each survey, the majority of plants were recorded as being in good condition.

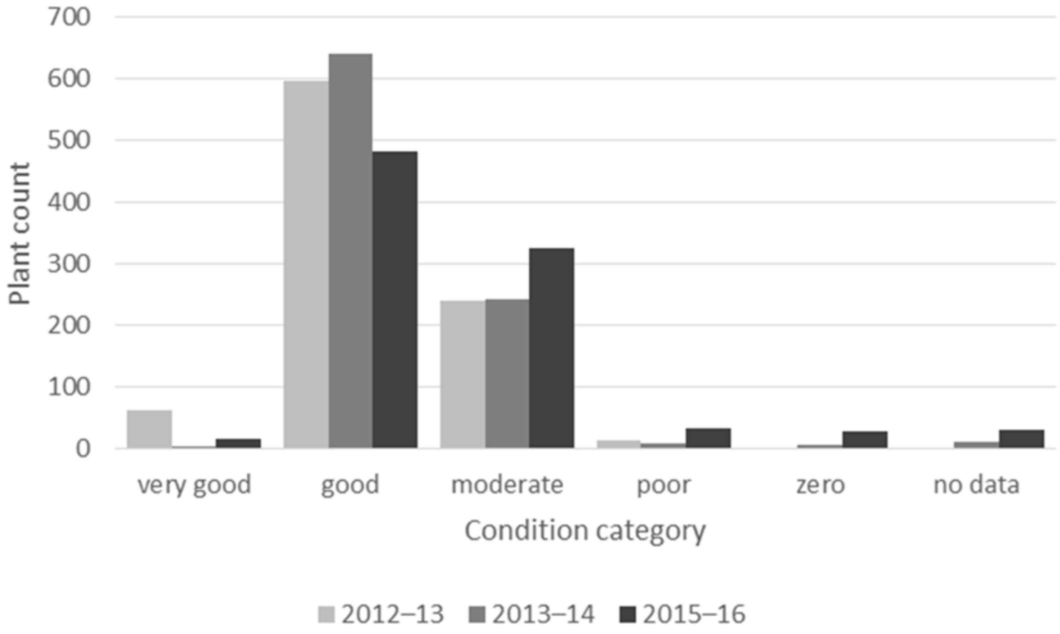


Figure 6-3. Count of LCI condition categories of all sites (including replacement plants and excluding discontinued plants) at LMW (n = 910 plants per survey).

Pooling plants for all condition categories, the mean LCI score was 7.495 in 2012–13, 7.220 in 2013–14 and 6.407 in 2015–16. There was a statistically significant difference between each successive pair of years (*P* = <0.001 for each pair of years, using ANOVA with Holm-Sidak pairwise comparisons).

6.5 Discussion

Over the last decade of TLM condition monitoring, the LMW icon site has experienced severe drought (mid 1990s to 2009), followed by widespread flooding (late 2010–11) (Leblanc et al. 2012). Higher-than-average rainfall in late 2010–11 across the Murray River catchment caused overbank flooding of the Murray River into LMW and the surrounding floodplain. Areas of floodplain that were not inundated by overbank flooding received substantial rainfall during the summer of 2010–11 (559 mm recorded compared with a long-term summer average of 135 mm at Lake Victoria) (BOM 2011). Despite this relief, the LMW icon site has experienced large-scale Lignum mortality over the last decade as a result of severe drought. In 2015–16, the number of plants recorded as zero (no viable above-ground biomass) increased to 253, which is more than half of the original 450 plants surveyed. The majority of these plants have shown no sign of regeneration following natural flooding and rainfall (summer 2010–11) and it is likely that they will not recover.

For Lignum communities to persist sustainably in the Murray–Darling Basin, plants require flood inundation approximately once in every three to 10 years (Craig et al. 1991; Rogers & Ralph 2011). Lignum can tolerate drought in a dormant state, regenerating from viable underground rootstock when environmental conditions become favourable (Brock et al. 2006). There are, however, limitations to the period of dormancy from which Lignum can recover. While dry periods may inhibit optimal Lignum growth, the known distribution of Lignum indicates that Lignum plants have the ability to respond to favourable environmental conditions after up to 10 years of inter-flood dry periods (Rogers & Ralph 2011). However, without a return to favourable conditions, plants lose the ability to regenerate and eventually perish (Roberts & Marston 2011). Some of the Lignum communities at LMW are located high on the floodplain and have not been inundated for more than 20 or 40 years. Increasing the power of the survey design by including more sites, and the analysis of the LCI in the last three surveys, suggests that Lignum condition is still declining. It is anticipated that a transition to the recommended survey methodology under the TLM refinements program will increase our understanding of the condition of Lignum communities across the LMW icon site in future monitoring years (Brown et al. 2015).

Lignum-dominated communities (e.g. EVCs such as Lignum swamp and Lignum shrubland) are a characteristic component of the LMW icon site (Brown et al. 2015). Lignum is the structurally dominant species in these communities, which encompass almost 10 000 ha at LMW. The site also supports an additional 10 000 ha of woodland communities in which Lignum is a key understorey species (e.g. EVCs such as intermittent swampy woodland, Lignum swampy woodland and shrubby riverine woodland) (Brown et al. 2015). The loss of Lignum, particularly from Lignum-dominated communities, has follow-on implications for the provision of habitat and other ecosystem functions. It is important to understand whether the increase in mortality of Lignum plants observed through TLM condition monitoring is widespread across the icon site or not.

6.5.1 Recommendations

There are several ways that we can improve our understanding of Lignum-dominated communities at the LMW icon site:

- Transition to the recommended methodology under the TLM refinements program (Brown et al. 2015)
- Undertake intervention monitoring of Lignum condition in areas where environmental flows can be delivered (such as Mulcra Island) to enable comparisons between the recovery potential and condition of Lignum communities after watering and at long-dry sites. This includes (but is not limited to) collating and analysing existing Lignum data that were collected for the Mallee CMA as part of a Mulcra intervention monitoring program. Five additional Lignum sites were established and surveyed in 2011 (prior to delivery of environmental water) and 2013 (following delivery of environmental water), specifically to capture Lignum response to environmental watering. It is likely that three of the five sites were inundated by environmental flows. Although these data were collected, they have not yet been analysed.
- Undertake additional investigations of Lignum condition and rootstock viability at LMW to better understand the extent of mortality and the potential for recovery. These investigations could include piloting (with CSIRO) the use of remote-sensing techniques to monitor the condition of Lignum-dominated communities across larger spatial scales.

6.5.2 Summary

Key points for Lignum condition monitoring at LMW in 2015–16 are:

- Overall, Lignum at LMW was in poor condition in 2015–16, largely due to the widespread mortality as a result of severe drought conditions experienced across the region (mid 1990s to 2009) and a continued lack of floodplain inundation. There has been an increase in the number of Lignum plants recorded as zero (dead or dormant) since 2010–11 to more than 50% of the 450 plants originally surveyed.
- Despite the widespread mortality, the majority of live Lignum plants surveyed were in good or moderate condition. It is anticipated that a transition to the recommended methodology under TLM refinements program will increase our understanding of the condition of Lignum communities across the LMW icon site in future monitoring years.
- Some of the Lignum communities at LMW are located high on the floodplain and have not been inundated for more than 20 or 40 years.

7 Cumbungi

BRAEDEN LAMPARD AND SCOTT HUNTLEY

7.1 Introduction

Cumbungi (*Typha spp.)* is an emergent plant that is wide spread across the Murray–Darling basin. Two species of Cumbungi occur at LMW, both of which are native: *T. orientalis* and *T. domingensis.* Cumbungi has a preference for stable water conditions, such as in weir pools, and for waterways with high nutrient levels. Under favourable conditions Cumbungi can become invasive and form high-density stands, which can hinder aquatic species movement and limit native vegetation growth (Roberts & Marston 2011). In large, dense stands it can be extremely hard to keep under control and can alter water flow in creeks and rivers (Flanery & Clark 2007). Cumbungi can also provide useful ecosystem services, for example, habitat for native wildlife, capture of silt and sediments and provision of nest sites for native wildlife (Chen et al. 2013).

7.2 Ecological objectives

The notion of Cumbungi monitoring came about after large, dense stands of Cumbungi were perceived as a potential problem for plant diversity, fish passage and flows, early in the development of the TLM Condition Monitoring Program. In Robinson's (2014a) review of condition monitoring programs this is specified as the adopted objective:

Limit Cumbungi growth.

This monitoring objective links to the overarching Environmental Water Management Plan (MDBA 2012b; MDBC 2007a) ecological objective for TLM Condition Monitoring Program:

Increase diversity and abundance of wetland aquatic vegetation.

The interpretation of this objective is that by limiting the dominating abundance of Cumbungi, other aquatic vegetation can increase in diversity and abundance.

For further information regarding the significance of Cumbungi as a monitoring component refer to MDFRC (2011).

7.3 Methods

7.3.1 Site information

Initial surveys of Cumbungi stands in anabranches and adjacent Murray River at LMW, were undertaken in 2006–07. The length of Cumbungi stands was measured as a total of 78 km across 10 reaches. To increase representation, in 2007–08 an additional reach was added (Lower Lock 6) and three of the existing reaches were extended to a total of 87.6 km across 11 reaches (Table 7.1).

Table 7.1. Length of reach surveyed at LMW and adjacent sections of the Murray River. Surveys were conducted annually from 2006–07 to 2015–16. Note: Mullaroo and Toupnein Creeks were not surveyed in 2010–11 due to inability to access the sites; Lower Lock 6 reach was added in 2007 and Lock 7 reach length was not calculated in 2006–07.

Location	Reach	Reach length surveyed (km)	
		2006–07	2007–08 to 2015–16
Lindsay	Lower Lindsay River	4.9	4.9
	Mullaroo Creek	8.6	8.6
	Toupnein Creek	7	7.3
	Upper Lindsay River	1.1	2.7
Mulcra	Potterwalkagee Creek	11.1	11.1
Wallpolla	Dedmans Creek	1	1
	Lower Wallpolla Creek	7.6	7.6
Murray River	Lower Lock 6	NA	9.6
	Upper Lock 6	17.5	17.5
	Lock 7	NA	8.2
	Lock 9	4.2	9.1

7.3.2 Sampling design

Each bank of the water course was surveyed and the location of Cumbungi stands recorded at each site using a hand-held GPS. A single stand is categorised as being > 1 m away from any other visible Cumbungi plant and Cumbungi was only measured if it was within 1 m of the bank's edge at normal weir pool height. A note was made if the stand of Cumbungi covered the full width of the channel. Depending on the size of the Cumbungi stand, the length of the stand was measured using one of three methods:

- with a tape measure if small (≤ 4 m)
- using a laser rangefinder (Opti-Logic 800XL/Bushnell Yardage Pro Sport 450) for stands of ≈4–500 m and within line of sight
- by taking a GPS point at the start, at any major directional changes and at the end of the stand and calculating the length using ArcGIS (v10.2) for very long stands.

All Cumbungi stands were recorded as *Typha* spp., with no distinction made between the two *Typha* species that occur at the LMW icon site. Surveys were conducted in spring/summer 2006–07, summer 2007–08, summer 2008–09, spring 2009–10, 2010–11, 2011–12, 2012–13, late winter 2013–14 and spring 2015–16. Monitoring did not occur in 2014–15 due to funding changes. Collection of data occurred at all reaches in all survey years, with the exception of Lower Lock 6 reach, where monitoring only began in 2007–08, and Mullaroo and Toupnein Creeks, where access was impossible during 2010–11 due to bridge closures in high river conditions.

Photo points were established at each reach in 2009–10. One photo point was established at each reach, except for Potterwalkagee Creek where three photo points were established due to its greater length and concentration of stands. Photos were retaken annually to 2015–16 (see section 6.2 of Part B of this report for photos).

7.3.3 Data interpretation

The total length of Cumbungi stands measured within each reach is used along with the total reach length to calculate the percentage of bank coverage by Cumbungi for each reach. This provides a valuable way to measure Cumbungi expansion between years at any given reach whilst also allowing us to identify reaches that are Cumbungi hot-spots.

As progress towards new objectives and targets for Cumbungi at LMW, it was decided to also report on the reaches grouped by their 'flow strata'. Each of the eleven reaches is designated a flow stratum, which corresponds to the flow-rate characteristics of the reach (slow flow anabranch, Murray River channel or fast flow anabranch; Table **7.2**)

The reaches are discussed with respect to the hydrology in the vicinity of each reach, with differences between reach hydrology a contributing factor to the results recorded.

Table 7.2**.** Reach categories for Cumbungi sites.

Flow stratum	**Slow flow anabranch**	**Murray River channel**	**Fast flow anabranch**
Reach	Lower Lindsay River	Lower Lock 6	Mullaroo Creek
	Upper Lindsay River	Upper Lock 6	Dedmans Creek
	Toupnein Creek	Lock 7	
	Lower Wallpolla Creek	Lock 9	
	Potterwalkagee Creek		

7.4 Results

During the first five years of TLM Cumbungi monitoring there was a general increase in total length of Cumbungi stands across all reaches (Table 7.3). Prior to flooding in 2010–11, the slow-flow reach category had favourable conditions for Cumbungi growth, allowing the highest percentage of bank cover of Cumbungi (Figure 7-1 to Figure 7-3).

In 2010–11, natural flooding led to a substantial reduction of Cumbungi stands, all but eliminating Cumbungi from LMW, except in Lower Wallpolla Creek. In the next two monitoring years, Lower Wallpolla Creek was the only site where Cumbungi stands were present (1.92% and 2.7% bank coverage in 2011–12 and 2012–13 respectively; Table 7.4).

In 2015–16, all survey reaches again contained Cumbungi. However, within most reaches the length of Cumbungi stands was much lower compared to before the natural flood in 2010–11. For example, Potterwalkagee Creek (slow flow anabranch), which had over 55% bank coverage in 2010–11, had less than 1% bank coverage in 2015–16 (Table 7.4).

Table 7.3. Total length of Cumbungi stands within reaches at LMW and adjacent sections of the Murray River surveyed annually from 2006–07 to 2015–16. Note: Mullaroo and Toupnein Creeks were not surveyed in 2010–11 as access was restricted by flooding, and Lower Lock 6 was only surveyed from 2007–08 onwards.

Location	Reach	Total length of Cumbungi (m)								
		2006–07	2007–08	2008–09	2009–10	2010–11	2011–12	2012–13	2013–14	2015-16
Lindsay	Lower Lindsay River	153	270	415	362	744	0	0	NA	65
	Mullaroo Creek	122	312	355	466	NA	0	0	NA	39
	Toupnein Creek	60	361	414	459	NA	0	0	NA	0.1
	Upper Lindsay River	545	1255	1702	2128	2455	0	0	NA	65
Mulcra	Potterwalkagee Creek	4898	10339	10057	11085	12315	0	0	NA	9
Wallpolla	Dedman's Creek	251	269	368	334	304	0	0	NA	16
	Lower Wallpolla Creek	1539	1888	2166	2372	3143	292	413	NA	1209
Murray River	Lower Lock 6	NA	555	878	711	989	0	0	NA	265
	Upper Lock 6	240	349	381	478	539	0	0	NA	6
	Lock 7	123	278	360	518	626	0	0	NA	208
	Lock 9	465	779	932	1058	1445	0	0	NA	216

Table 7.4. Percent bank cover of Cumbungi stands at LMW. Note: Mullaroo and Toupnein Creeks were not surveyed in 2010–11 as access was restricted by flooding, and Lower Lock 6 was only surveyed from 2007–08 onwards.

Location	Reach	% of bank covered								
		2006–07	2007–08	2008–09	2009–10	2010–11	2011–12	2012–13	2013–14	2015-16
Lindsay	Lower Lindsay River	2	3	4	4	8	0	0	NA	1
	Mullaroo Creek	1	2	2	3	NA	0	0	NA	0
	Toupnein Creek	0	2	3	3	NA	0	0	NA	0
	Upper Lindsay River	25	23	32	39	46	0	0	NA	1
Mulcra	Potterwalkagee Creek	22	47	45	50	56	0	0	NA	0
Wallpolla	Dedman's Creek	13	13	18	17	15	0	0	NA	1
	Lower Wallpolla Creek	10	12	14	16	21	2	3	NA	8
Murray River	Lower Lock 6	NA	3	5	4	5	0	0	NA	1
	Upper Lock 6	1	1	1	1	2	0	0	NA	0
	Lock 7	NA	2	2	3	4	0	0	NA	1
	Lock 9	6	4	5	6	8	0	0	NA	1

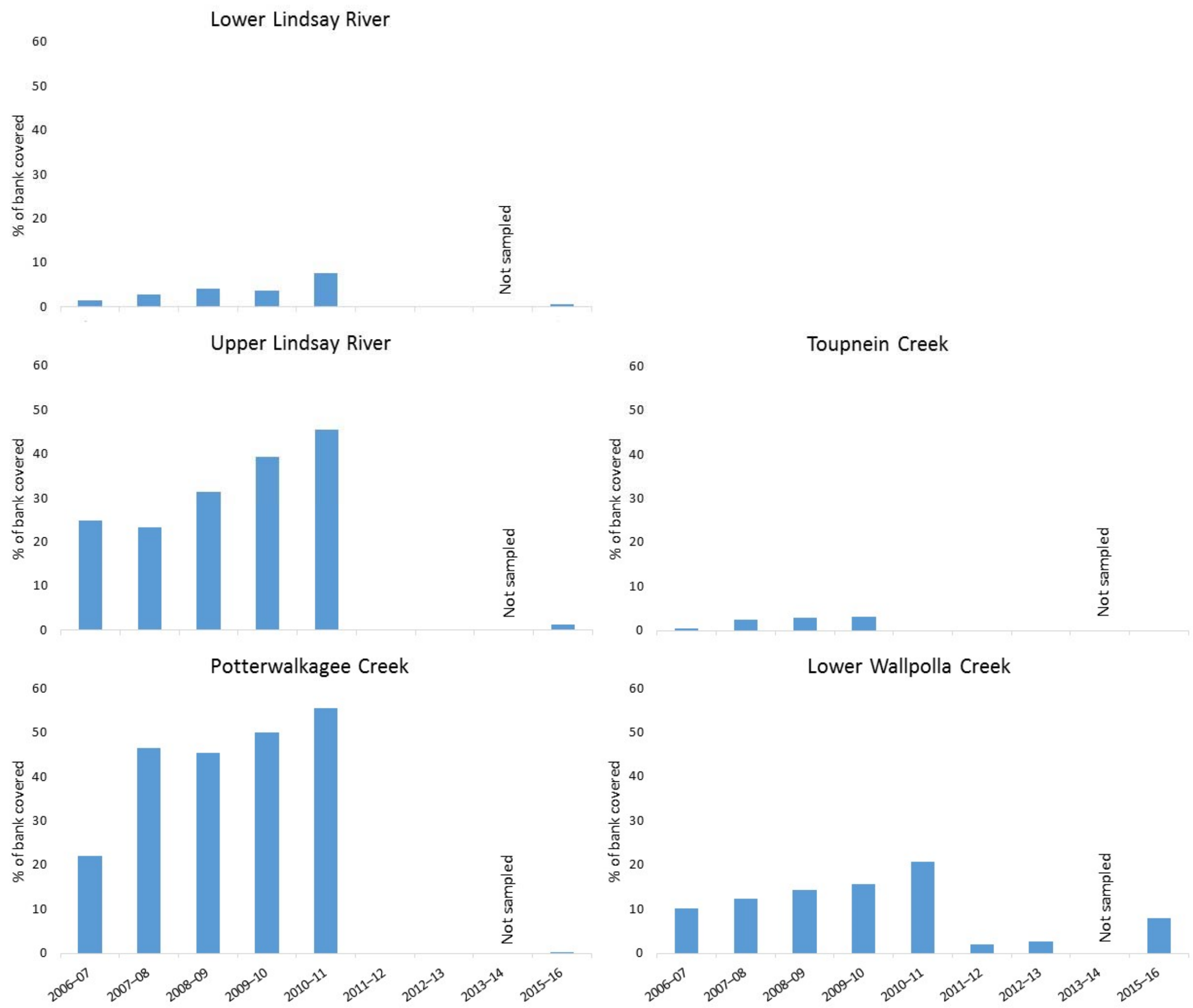


Figure 7-1. Percentage bank coverage of Cumbungi across all monitoring years for the slow flow anabranch stratum.

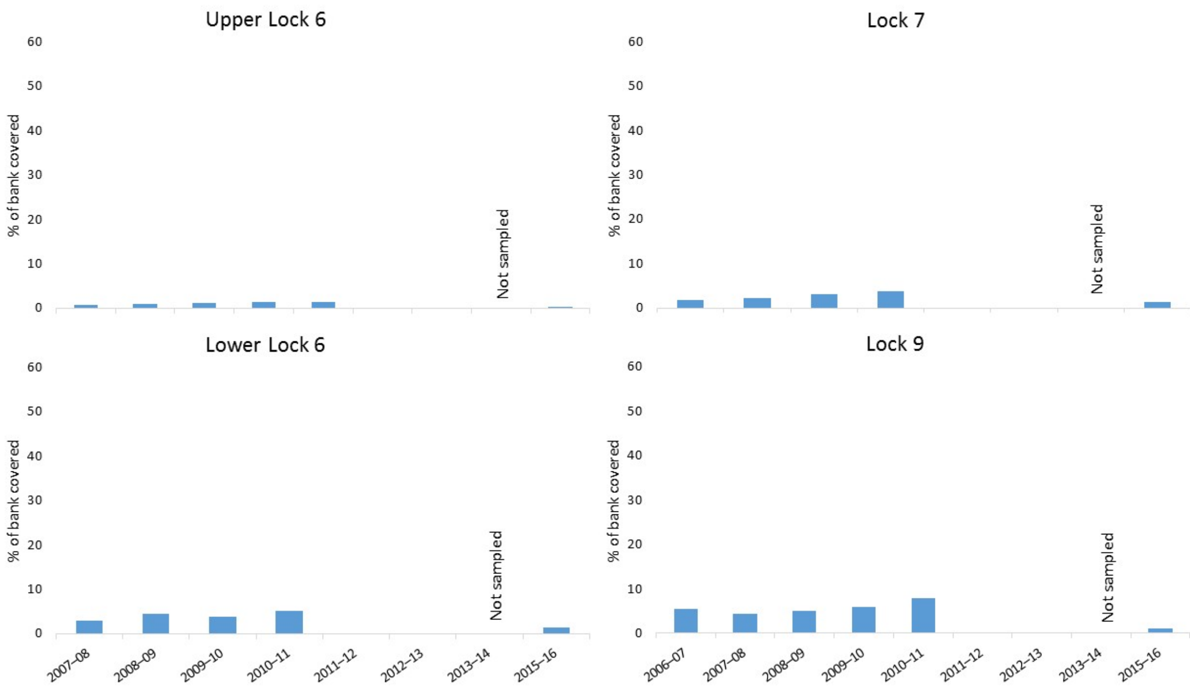


Figure 7-2. Percentage bank coverage of Cumbungi across all monitoring years for the Murray River channel stratum.

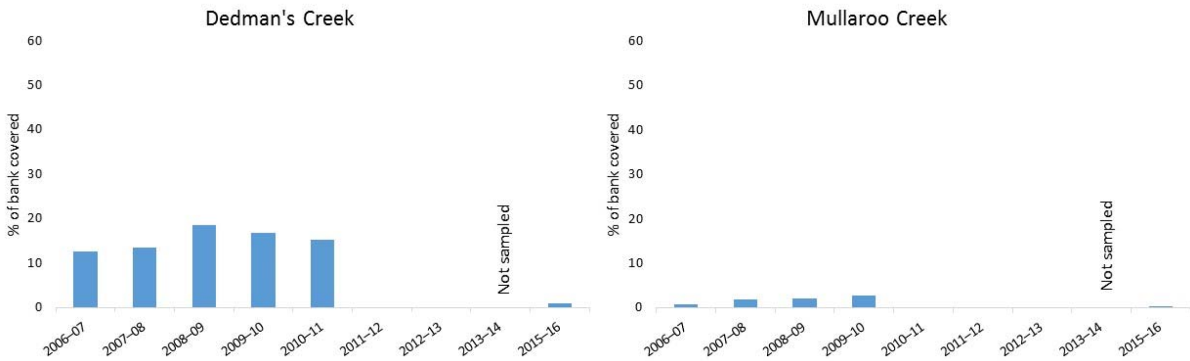


Figure 7-3. Percentage bank coverage of Cumbungi across all monitoring years for the fast flow stratum.

7.5 Discussion

Before the 2010–11 flood, Cumbungi was widely distributed across LMW. Abundance was particularly high at slow-flow sites, with up to 55% bank coverage recorded. At almost all sites, Cumbungi stand length and percentage of bank coverage had been steadily increasing since the beginning of the monitoring program. The natural flooding in 2010–11 caused water levels across the Murray–Darling Basin, including at LMW, to rise between 1 and 4 m above weir-pool levels. This caused Cumbungi stands across LMW to be fully submerged for a long period, which starved plants of certain resources (light and oxygen) that they need to survive (Flanery & Clark 2007). After the initial peak of the flood, higher-than-average rainfall resulted in continued high water levels across most reaches in LMW. This led to a wide-scale disappearance of Cumbungi biomass across all reaches within LMW except Lower Wallpolla Creek. The influence of weir-pool protected Lower Wallpolla Creek from high water levels, allowing Cumbungi to persist there, although there was a more than 10-fold reduction in bank coverage.

For the two survey years subsequent to the 2010–11 flood, Cumbungi coverage was zero in all reaches except Lower Wallpolla Creek. However, in the 2015–16 monitoring period, Cumbungi was observed to have re-established in all reaches, although abundance is still substantially lower than pre-flood levels.

Lower Lock 6, Lock 7 and Lock 9, Lower Lindsay and Lower Wallpolla have shown greater increases in the percentage of bank coverage and total length of Cumbungi stands compared to the other reaches in LMW in the 2015–16 monitoring season. These reaches are influenced by weir pools and therefore have relatively stable water levels.

Compared to 2010–11, Potterwalkagee Creek showed the largest reduction in Cumbungi bank cover in the 2015–16 survey. Since 2010, a number of block banks, weirs, and regulators have been built in this creek, which has resulted in unfavourable conditions for Cumbungi to re-establish (wet–dry cycles). Before the construction of these structures, stable water levels at this site created ideal conditions for Cumbungi to flourish (warm, still or low-flowing water Flanery & Clark (2007).

7.5.1 Progress towards objectives

Current ecological objectives for limiting the extent and colonisation of Cumbungi are designed to support the objective of increasing the diversity and abundance of wetland aquatic vegetation, because Cumbungi can rapidly colonise and out-compete other species.

Although Cumbungi cover is currently increasing at LMW, it has not yet reached the extent and cover experienced in 2009–10, which was considered indicative of poor condition for other wetland vegetation species (Robinson 2014a). The current level of cover is not yet considered detrimental to other wetland species and therefore the adopted objective *Limit Cumbungi growth* is being met. However there is no evidence yet that growth rates are reduced, or that populations have stabilised through effective management strategies to keep Cumbungi in check.

7.5.2 Recommendations

Currently, there is no specific ecological objective for Cumbungi extent at LMW; it is recommended that an objective should be developed. “Good” and “poor” condition reference points need to be developed to ensure accurate reporting against objectives, as currently it is difficult to determine at what point Cumbungi extent is beyond a critical point that is detrimental to competing wetland vegetation species. Robinson (2014a) suggests that “good” and “poor” condition should be defined as “extent = 0%” and “extent in 2009”, respectively. However, in ecological terms an absolute lack of any endemic aquatic vegetation, Cumbungi included, is probably not a valuable aim. In moderate extent, stands of Cumbungi can provide useful habitat for native wildlife, capture of silt and sediments and provide nest sites for native wild life (Chen et al. 2013). Also, the peak of Cumbungi extent was actually surveyed in 2010, prior to the high flows that inundated the stands. A useful reference level condition should include moderate extent of Cumbungi within at least some of the reaches monitored.

8 Fish

SCOTT HUNTLEY, DAVID WOOD AND 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 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 has resulted in significant change from the natural flow regime and it is generally believed that (at least) large-bodied 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. Planned works have been completed in the Lindsay River, Mullaroo Creek, Lake Wallawalla, Websters Lagoon and Potterwalkagee Creek 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 resulting from the implementation of works programs and the application of environmental water. Monitoring is specifically tailored to determine if management objectives are being met.

8.2 Ecological objectives and indices

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 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 is suggested and a suite of indices were investigated (Robinson 2014a):

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

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

- α-species diversity index
  - within-habitat diversity, where habitat refers to macrohabitat (riverine, anabranch, channel, wetland)
- β-species diversity index
  - inter-habitat diversity, where habitat refers to macrohabitat (riverine, anabranch, channel, wetland)
- expectedness index
  - the average observed-to-expected score for each reach/wetland within each year for each macrohabitat
- nativeness index
  - the percent biomass of native compared to all fish
- extent index
  - the percentage of in which sites each species occurred
  - calculated for each macrohabitat as the average across all species.

8.3 Methods

8.3.1 Sampling

Sampling for fish under the TLM Condition Monitoring Program at LMW has occurred on nine occasions: spring (September–November) 2006 (electrofishing only), spring (September–November) 2007 (except for Toupnein Creek, which was sampled in early January 2008), spring (September) 2009, and the autumns (March–April) of 2010, 2011, 2012, 2013, 2014 and 2016. Sampling events are, for consistency, hereafter referred to as the 2006–07, 2007–08, 2008–09, 2009–10, 2010–11, 2011–12, 2012–13, 2013–14 and 2015–16 monitoring seasons or years. The wetland macrohabitat of Websters Lagoon was added to the program in 2010–11, and then sampled again in 2011–12, 2012–13 and 2013–14. Websters Lagoon was dry in 2015–16 and couldn't be sampled.

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 Dedmans 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 section 10 of MDFRC (2011).

Table 8.1. Location of fish sampling sites within each macrohabitat and reach.

<table>
<tr><th>Macrohabitat</th><th>Location</th><th>Reach</th><th colspan="3">Site codes</th></tr>
<tr><td rowspan="4">Riverine</td><td rowspan="4">Murray River</td><td>Lock 6 weir pool (L6)</td><td>L6.1</td><td>L6.2</td><td>L6.3</td></tr>
<tr><td>Lock 7 weir pool (L7)</td><td>L7.1</td><td>L7.2</td><td>L7.3</td></tr>
<tr><td>Lock 8 weir pool (L8)</td><td>L8.1</td><td>L8.2</td><td>L8.3</td></tr>
<tr><td>Lock 9 weir pool (L9)</td><td>L9.1</td><td>L9.2</td><td>L9.3</td></tr>
<tr><td rowspan="5">Anabranch (no/slow flow)</td><td rowspan="3">Lindsay</td><td>Lower Lindsay River (LLR)</td><td>LLR.1</td><td>LLR.2</td><td>LLR.3</td></tr>
<tr><td>Upper Lindsay River (ULR)</td><td>ULR.1</td><td>ULR.2</td><td>ULR.3</td></tr>
<tr><td>Toupnein Creek (TC)</td><td>TC.1</td><td>TC.2</td><td>TC.3</td></tr>
<tr><td>Mulcra</td><td>Potterwalkagee Creek (PC)</td><td>PC.1</td><td>PC.2</td><td>PC.3</td></tr>
<tr><td rowspan="2">Wallpolla</td><td>Lower Wallpolla Creek (LWC)</td><td>WMC.1</td><td>WMC.2</td><td>WMC.3</td></tr>
<tr><td rowspan="2">Channel (fast flow anabranch)</td><td>Dedmans Creek (DC)</td><td>DC.1</td><td>DC.2</td><td>–</td></tr>
<tr><td rowspan="2">Lindsay</td><td>Mullaroo Creek (MC)</td><td>MC.1</td><td>MC.2</td><td>MC.3</td></tr>
<tr><td>Wetland</td><td>Websters Lagoon (WL)</td><td>WL.1</td><td>WL.2</td><td>WL.3</td></tr>
</table>

Sampling limitations

A limitation with fish sampling is that no single technique is effective for all species or for all age cohorts of individual species (Murphy & Willis 1996). The range of sampling techniques employed in this study has been developed to provide an efficient and, where possible, consistent method for sampling a range of species with diverse behaviours and habitat preferences across multiple sites throughout the Murray River and its associated waters (MDBA 2010). The methods are considered a good compromise between sampling effort and representativeness; however, there are a suite of factors that affect gear efficiency and selectivity that need to be appreciated.

Electrofishing efficiency is dependent on a suite of biological, environmental and technical factors. Biological factors, such as morphology, physiology and behaviour (Reynolds 1996), may make large edge-dwelling species such as Common carp (*Cyprinus carpio*), Golden perch (*Macquaria ambigua*) and Murray cod (*Maccullochella* peelii) more susceptible to capture by electrofishing than small-bodied fish and fish that inhabit benthic (e.g. Freshwater catfish, *Tandanus tandanus*) or pelagic habitats (e.g. Bony bream, *Nematalosa erebi*). Environmental factors influencing electrofishing efficiency include salinity, temperature, turbidity, dissolved oxygen, morphology, substrate, underwater structure (e.g. snags, vegetation), weather and time of day (Reynolds 1996). Electrofishing is less effective in water with very low or very high salinity, where the required power density (power transfer) to evoke a fish response may be unattainable (Kolz 2006). Salinity levels too high for electrofishing occur periodically in the upper Lindsay River. Where sampling cannot be conducted using electrofishing, large fyke nets are deployed to sample large-bodied fish.

Unbaited bait trap sampling has been largely ineffective at sampling small-bodied fish at LMW (MDFRC, unpublished data) and has therefore been complemented by small fyke net (SFN) and seine net sampling. Small fyke netting can be selective for species and size, whilst efficiency can be affected by abiotic factors such as season, water temperature, time of day, water level fluctuation, turbidity and currents (Hubert 1996; Laarman & Ryckman 1982). Seine nets are most effective on fishes that tend to swim in front of the net (rather than to the sides or underneath (Hayes et al. 1996), and those which inhabit the middle of the water column of littoral habitats (Lyons 1986) and around particular macrophyte beds (Allen et al. 1992). Seine efficiency can be significantly compromised by bottom structure such as the presence of large rocks, ledges and woody debris (Hayes et al. 1996). Seine netting was therefore undertaken in areas free of obstructions that may have interfered with the gear.

In order to minimise sample variance associated with gear selectivity, a number of standardisation controls were employed including: standard operating procedures, consistency in sampling season, consistency in sampling sites and gear type. It is therefore possible to make confident assertions about changes in the fish community between years based on catch differences but with less confidence about differences between the abundance of different species.

8.3.2 Statistical analyses

Species diversity index

The species diversity index is comprised of two indices used to investigate changes in native species diversity between years: (i) intra-macrohabitat (α; alpha) diversity and (ii) inter-macrohabitat (β; beta) diversity.

Intra-macrohabitat diversity (α-diversity) was calculated as the average (mean) number of native species recorded at each reach within each macrohabitat. Species count was averaged for sites to produce reach averages, which were then averaged to produce an α-diversity (± 95% CI) value for each macrohabitat.

Inter-macrohabitat diversity (β-diversity) was calculated as the total number of native species present within each macrohabitat in a given year. Therein, β-diversity is the native species richness for each macrohabitat for each year.

Expectedness index

The expectedness index is a measure of how many species occurred year to year compared to how many were expected to occur based on how rare and how catchable a species is at the given location. The expectedness index is based on that used as part of the Sustainable Rivers Audit (MDBC 2007b), where each species was assigned a rarity score (RC-F) of 0, 1, 3 or 5 based on expert opinion (Table 8.2) then assigned a corresponding expectedness weight (ωk). However, because there is considerable TLM monitoring data available on which to establish RC-F and ωk values for the LMW icon site, in the current report frequency-of-detection-based expectedness values were used instead of expert opinion.

The premise of this method is that each icon site has a list of potential species that can occur, generated from the data that have been sampled over the nine years of the TLM program. For example, if the data show that a species of fish occurred in an average of seven out of nine wetlands over the nine years of the TLM condition monitoring data set, then it may be reasonable to expect that species to occur in 78% of sites sampled in future surveys. Each species is then given a RC-F value of 1, 3 or 5 depending on the percentage of that occurrence (see Table 8.2) based on the system originally determined for the Sustainable Rivers Audit (MDBC 2007b).

So in the example, if a species occurs in 78% of sites it would receive a RC-F value of 5, meaning it is a common and abundant species. Expectedness metric weight (ωk) is then derived from the RC-F value, given factors affecting detection, such as rareness, catchability (electrofishing susceptibility, net avoidance etc.), body size, etc. In the example the RC-F value of 5 would be weighted as 0.85 expectedness. This is to account for the premise that not every species should be expected to be collected in every site but that most species should be collected somewhere in the macrohabitat (Robinson 2013).

Table 8.2. The RC-F fish value interpretation and associated expectedness weighting.

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

Nativeness index

Most fish were measured and weighed at capture. Some fish were unable to be accurately weighed in the field. For those fish that were measured (L mm, standard length) but not weighed, we estimated the weight ($\overline{W}$, grams) based on:

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

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, the first 50 individuals were measured and weighed as they were encountered in the first replicates. The remainder, in subsequent replicates were simply counted, therefore in a minority of replicates there were no measured individuals of a sampled species.

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

For *pNative* the lowest level of replication built into the survey design was at the 'gear' level. Replicate sub-samples were taken using multiple nets, or electrofishing 'shots' at each site. High catch variability among these sub-samples was accounted for in analysis of variance by partitioning the variance down to this level when calculating the variance for mean *pNative* for each macrohabitat within the icon site. We calculated the standard errors directly from the mean square errors from the nested ANOVA for each year, using the formula from Nichols et al. (2006) where the model is:

$$pNative = \text{Reach} \times \text{Reach(Site)} + error$$

where the replicates are the nets or electrofishing shots and sites are nested within reach.

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

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

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

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

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

Extent index

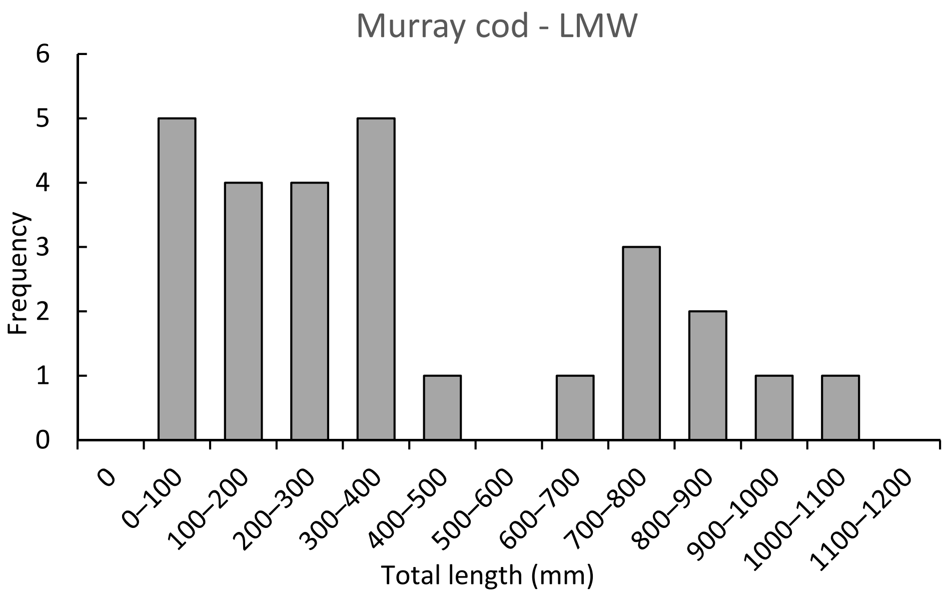
The extent index is a measure of how frequently each native fish species occurs within a particular macrohabitat. The extent index is calculated as the average (± 95% CI) number of native species sampled within each site as a proportion of the maximum possible number of species that could be sampled (i.e. all species at all sites) to produce a score for each sampling period (year) between 0–1.

8.4 Results

In total, 34 888 fish were sampled from 15 species (11 native and 4 non-native) at LMW Islands in the current survey of 2015–16 (Table 8.3). This is the largest catch of fish at LMW for all monitoring years. The next-largest catch was recorded in the 2010–11 monitoring year, which was a flood year.

Numbers of Murray cod sampled in 2015–16 were the highest recorded since the pre-flood monitoring year (2008–09). Golden perch numbers were the highest recorded since 2010–11 and 2011–12 monitoring years (flood and high-flow monitoring years). Population distributions of Murray cod and Golden perch across all macrohabitats are presented in Figure 8-1 for the 2015–16 monitoring year.

Websters Lagoon, the only representative for the wetland macrohabitat, was not sampled during the 2015–16 monitoring year due to insufficient water level. Analysis was therefore not conducted for this macrohabitat for this monitoring year.



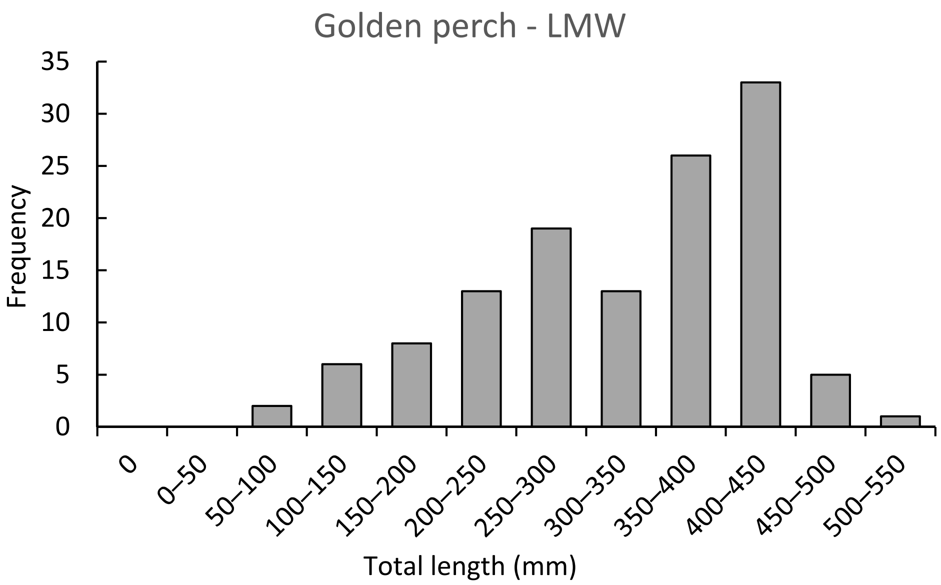


Figure 8-1. Length–frequency histogram for Murray cod (top) and Golden perch (bottom) across all macrohabitats at LMW during the 2015–16 monitoring season.

Table 8.3. Summary counts of all fish sampled at LMW Islands over nine sampling years as part of TLM Condition Monitoring Program. Note: 2014–15 not sampled.

		Native													Non-native							
		Large-bodied						Small-bodied							Large-bodied					Small-bodied		
Year	Habitat	Bony herring	Freshwater catfish	Golden perch	Murray cod	Silver perch	Spangled perch	Australian smelt	Carp gudgeon	Dwarf flathead gudgeon	Flathead gudgeon	Murray-Darling rainbowfish	Un-specked hardyhead	Native Total	Common carp	Common carp x Goldfish	European perch	Goldfish	Oriental weatherloach	Eastern mosquitofish	Non-native Total	Grand Total
2006–07	Anabranch	190		41		3								234	150		39	34			223	457
	Channel	13		18	9	1								41	22		1	5			28	69
	Riverine	91		41	13	3								148	95		6	14			115	263
	Wetland	Not Sampled																				
	Total	294		100	22	7								423	267		46	53			366	789
2007–08	Anabranch	475		42	1	5		44	3186	2	50	12	987	4804	132		14	71		1659	1876	6680
	Channel	148		20	43	4		151	724		18	3	194	1305	54		1	1		251	307	1612
	Riverine	225		62	29	3		112	1351		294	28	818	2922	102		9	9		1483	1603	4525
	Wetland	Not Sampled																				
	Total	848		124	73	12		307	5261	2	362	43	1999	9031	288		24	81		3393	3786	12817
2008–09	Anabranch	8		16	8	4		179	3189	15	130	43	533	4125	72		5	48		451	576	4701
	Channel			11	36	8		98	367	2	15	3	33	534	38			1		305	344	917
	Riverine	18		20	5			117	2435	1	565	39	1563	4763	42		3			336	381	5144
	Wetland	Not Sampled																				
	Total	26		47	49	12		394	5991	18	710	85	2129	9461	152		8	49		1092	1301	10762
2009–10	Anabranch	2162		18	2	2		68	2630	14	42	18	529	5485	154		9	140		1681	1984	7469
	Channel	288		10	16	1		74	128	1	4	4	80	606	38			11		197	246	852
	Riverine	4165		78	9	3		279	759	6	155	20	756	6230	207		22	179		738	1146	7376
	Wetland	Not Sampled																				
	Total	6615		106	27	6		421	3517	21	201	42	1365	12321	399		31	330		2616	3376	15697
2010–11	Anabranch	594		59		4	1	40	439	2	7	17	321	1484	764			113		10646	11523	13007
	Channel	168		24	1	2	1	5	17		1	2	8	229	128			12	1	1840	1981	2210
	Riverine	506	1	92	3	4	1	14	232	4	42	38	93	1030	556	1	4	44	1	9213	9819	10849
	Wetland	20		22					24		3		38	107	103		2	3		1324	1432	1539
	Total	1288	1	197	4	10	3	59	712	6	53	57	460	2850	1551	1	6	172	2	23023	24755	27605
2011–12	Anabranch	1698	5	68	1		2	27	659		440	2	11	2913	1002			93		17	1112	4025
	Channel	322	2	32	1		1	35	6		7	2	1	409	229			8		5	242	651
	Riverine	1290	8	74		1		215	259		207	16	3	2073	611		4	30		8	653	2726
	Wetland	20	1	10				2	140		80		7	260	156		2	1		55	214	474
	Total	3330	16	184	2	1	3	279	1064		734	20	22	5655	1998		6	132		85	2221	7876
2012–13	Anabranch	707	1	45				99	2355		347	1	29	3584	255		1	10		214	480	4064
	Channel	1415	1	13		1		3	487		3	21	10	1954	73					340	413	2367
	Riverine	1626		39	3	4		278	1076		898	18	62	4004	272		1	3		656	932	4936
	Wetland	20						1	32		34			87	26					54	80	167
	Total	3768	2	97	3	5		381	3950		1282	40	101	9629	626		2	13		1264	1905	11534
2013–14	Anabranch	2048		53				79	3450	8	217	33	18	5906	381			129		723	1233	7139
	Channel	2450	3	26	8	4	1	208	757		6	107	11	3581	74			4		66	144	3725
	Riverine	4049		50	6	2	3	305	1747		423	185	45	6815	229			19		262	510	7325
	Wetland	1							4		4			9	7					209	216	225
	Total	8548	3	129	14	6	4	592	5958	8	650	325	74	16311	691			152		1260	2103	18414
2014–15		Not Sampled																				
2015–16	Anabranch	3452		52		2		848	6426	6	1430	301	2655	15172	254		1	218		1418	1891	17063
	Channel	1517	3	30	21	2		757	1395	4	8	93	757	4587	75			48		280	403	4990
	Riverine	3201		78	12	9		892	3804	2	1970	405	967	11340	179			71		1245	1495	12835
	Wetland	Not Sampled																				
	Total	8170	3	160	33	13		2497	11625	12	3408	799	4379	31099	508		1	337		2943	3789	34888

8.4.1 Alpha species diversity

Alpha species diversity fluctuated across years in all macrohabitats (Figure 8-2). For the anabranch macrohabitat, α-diversity was significantly lower in 2012–13 than in most other years. This is primarily as a result of Australian smelt (*Retropinna semoni*), Dwarf flathead gudgeon (*Philypnodon macrostomus*), Flathead gudgeon (*Philypnodon grandiceps*) and, to a lesser extent, Murray–Darling rainbowfish (*Melanotaenia fluviatilis*) and Un-specked hardyhead (*Craterocephalus fulvus*) being recorded less frequently in that year. Anabranch α-diversity in 2015–16 was significantly higher than in most other years due to the majority of species occurring in each reach being recorded at each site within that reach.

Riverine α-diversity was significantly lower than average in 2008–09 and 2011–12. In 2008–09, this was associated predominately with Bony herring being encountered less often but also with lower-than-average recordings of Golden perch, Silver perch (*Bidyanus bidyanus*) and Murray–Darling rainbowfish at sites that year. The lower riverine α-diversity in 2011–12 was associated with a smaller representation of Un-specked hardyhead, Murray–Darling rainbowfish, Flathead gudgeon and Australian smelt within sites. Riverine α-diversity in 2015–16 was significantly higher than in most other years due to the majority of species occurring in each reach being recorded at each site within that reach.

There were no significant differences between years in the channel macrohabitat with the exception of the current monitoring year (2015–16). This is in part an artefact of the current sampling design, which provides a relatively low level of representation for this macrohabitat (only five sites nested within two channel habitats). Again, the difference between 2015–16 and some of the other years is due to the majority of species occurring in each reach being recorded at each site within that reach.

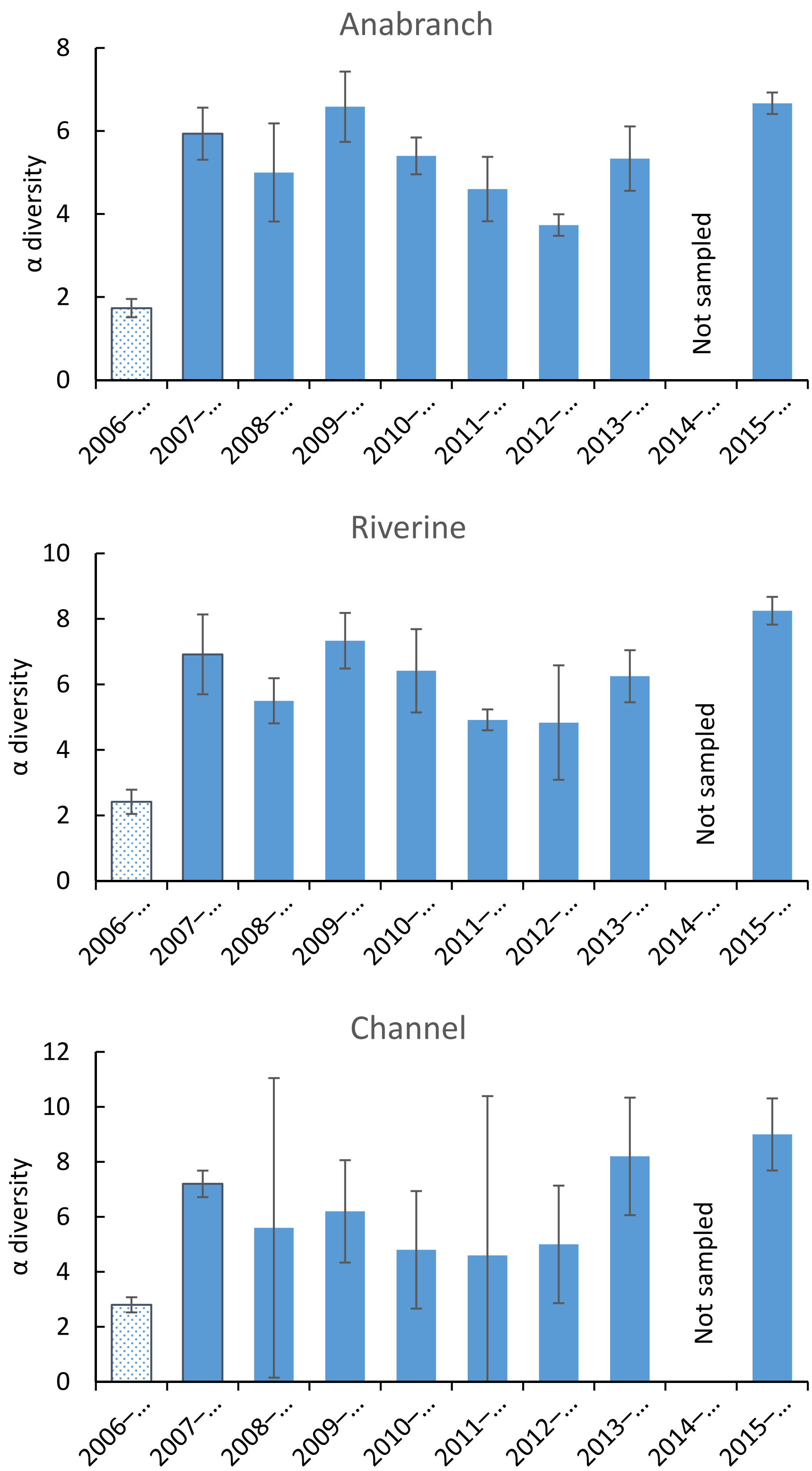


Figure 8-2. Mean (± 95% CI) α-diversity (within-macrohabitat diversity) of native species at LMW Islands between 2006–07 and 2015–16.

Note: small fyke nets were included in the sampling design from 2007–08 (i.e. not used in 2006–07; dotted bars).

8.4.2 Beta species diversity

Beta species diversity (Figure 8-3) was greatest in the riverine habitat in 2010–11 (flood year) due to the presence of all species, including Freshwater catfish and Spangled perch (*Leiopotherapon unicolor*), in that year. This was the first occurrence of Spangled perch at LMW in TLM condition monitoring. Channel β-diversity was highest in the most recent years (2013–14 and 2015–16) with 11 of the 12 previously sampled native species recorded, the exception being Dwarf flathead gudgeon which, whenever sampled, are generally recorded in very low numbers. Anabranch β-diversity was relatively consistent; however, the assemblage of species varied slightly between years. Species sampled not sampled every year in anabranch habitat were Dwarf flathead gudgeon, Freshwater catfish, Murray cod, Silver perch and Spangled perch. Across all macrohabitats the β-diversity is equal to or has increased from the previous monitoring survey in 2012–13.

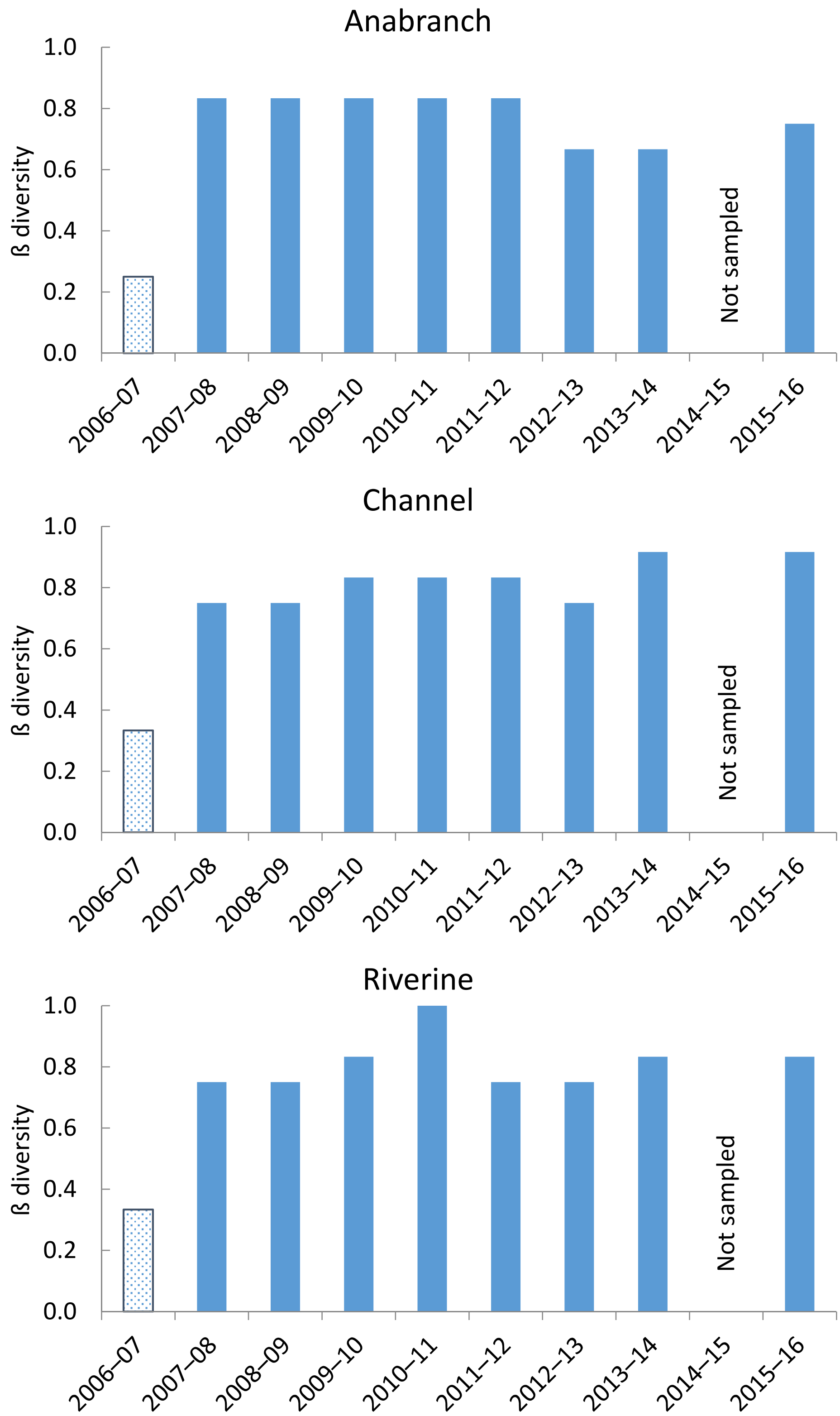


Figure 8-3. β-diversity index scores (macrohabitat species diversity: sampled vs optimum) for native fish species at LMW Islands between 2006–07 and 2015–16.

Note: small fyke nets were included in the sampling design from 2007–08 (i.e. not used in 2006–07; dotted bars).

8.4.3 *Expectedness*

The average number of species recorded at each site sampled from 2006–07 to 2015–16 trended similarly across macrohabitats (Figure 8-4). For most years, the number of species per site was less than expected $\Sigma(\omega k_j)$. The most notable exception was for the channel macrohabitat in 2013–14 and 2015–16, where an average of 8.1 (± 1.6) and 9.1 (± 1.2) species were sampled at each site, compared to an expected 6.7 species. This was also the case in the riverine macrohabitat for 2007–08, 2009–10 and 2015–16, where an average of 6.92 (± 0.8), 7.33 (± 0.7) and 8.25 (± 0.6) species were sampled at each site, compared to an expected 6.7 species. Based on pairwise comparisons of 95% CIs, average species counts were close to or met expectations in the anabranch macrohabitat in 2007–08, 2009–10 and 2015–16, in the channel macrohabitat in 2007-08, 2008–09, 2009-10 and 2011-12, and in the riverine macrohabitat in 2007-08, 2009–10, 2010–11 and 2013–14.

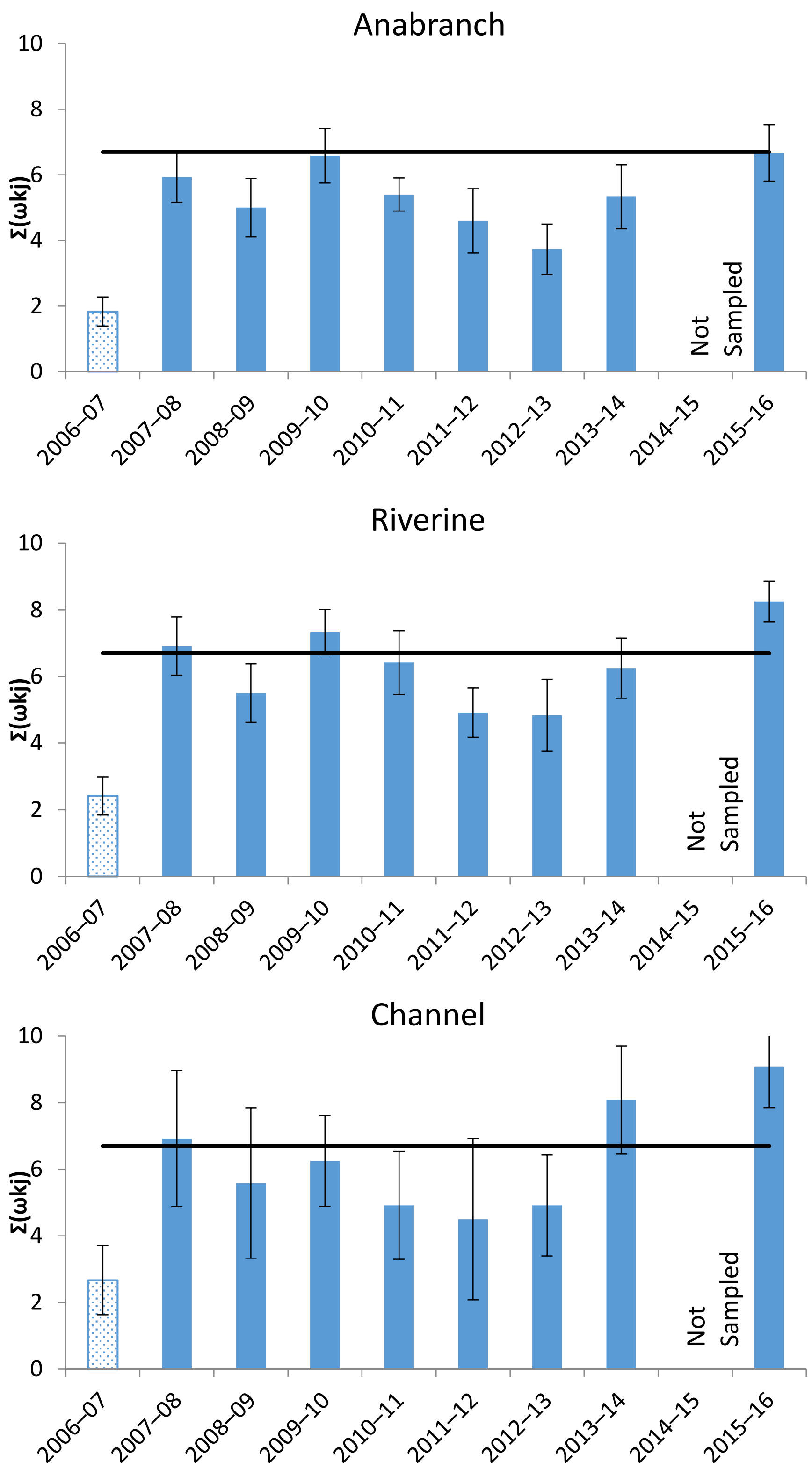


Figure 8-4. Average (mean ± 95% CI) number of native species recorded per site for each macrohabitat with respect to the expected number of species (expectedness reference $\Sigma(\omega k_j)$; horizontal line) calculated from historical data.

Note: small fyke nets were included in the sampling design from 2007–08 (i.e. not used in 2006–07; dotted bars).

8.4.4 Nativeness

Mean annual *pNative* (± SE) by gear type (small fyke nets, electrofishing boat, seine net and bait traps) and macrohabitat (anabranch, channel, riverine and wetland) are shown in Figure 8-5 and Figure 8-6. Estimates of these annual means for most *pNative* data, contain a high level of uncertainty due to variability among reaches, sites and gear-replicates.

In anabranch macrohabitats, the proportion of native fish biomass (*pNative*) varied significantly among years for data collected using small fyke nets (χ^2(df = 7) = 91, $p < 0.001$), seine nets (χ^2(df = 7) = 60, $p < 0.001$) and boat electrofishing (χ^2(df = 8) = 71, $p < 0.001$). Pairwise T-tests showed that, while there was no change to 2015–16 since the last survey (2013-14), there was a significant increase in *pNative* since minima in 2010–11 and 2011–12 ($p = 0.002$), Figure 8-5 and Figure 8-6.

In channel macrohabitats, *pNative* varied significantly among years for data collected using small fyke nets (χ^2(df = 7) = 56, $p < 0.001$), seine nets (χ^2(df = 7) = 21, $p = 0.004$) and boat electrofishing (χ^2(df = 8) = 46, $p < 0.001$). Pairwise T-tests showed that there was again no significant increase to 2015–16 since the last survey; however, *pNative* was significantly higher for small-fyke-net data ($p < 0.001$) and boat-electrofishing data ($p = 0.03$) in 2015–16 than it was in 2010–11 (Figure 8-5).

In riverine macrohabitats, *pNative* also varied significantly among years for data collected using small fyke nets (χ^2(df = 7) = 49, $p < 0.001$), seine nets (χ^2(df = 7) = 65, $p < 0.001$) and boat electrofishing (χ^2(df = 8) = 130, $p < 0.001$). Pairwise T-tests showed that there was again no significant increase to 2015–16 since the last survey; however, *pNative* was significantly higher for small-fyke-net data ($p < 0.001$), seine-net data ($p < 0.001$) and boat-electrofishing data ($p = 0.03$) in 2015–16 than it was in 2010–11 and 2011–12 (Figure 8-5 and Figure 8-6).

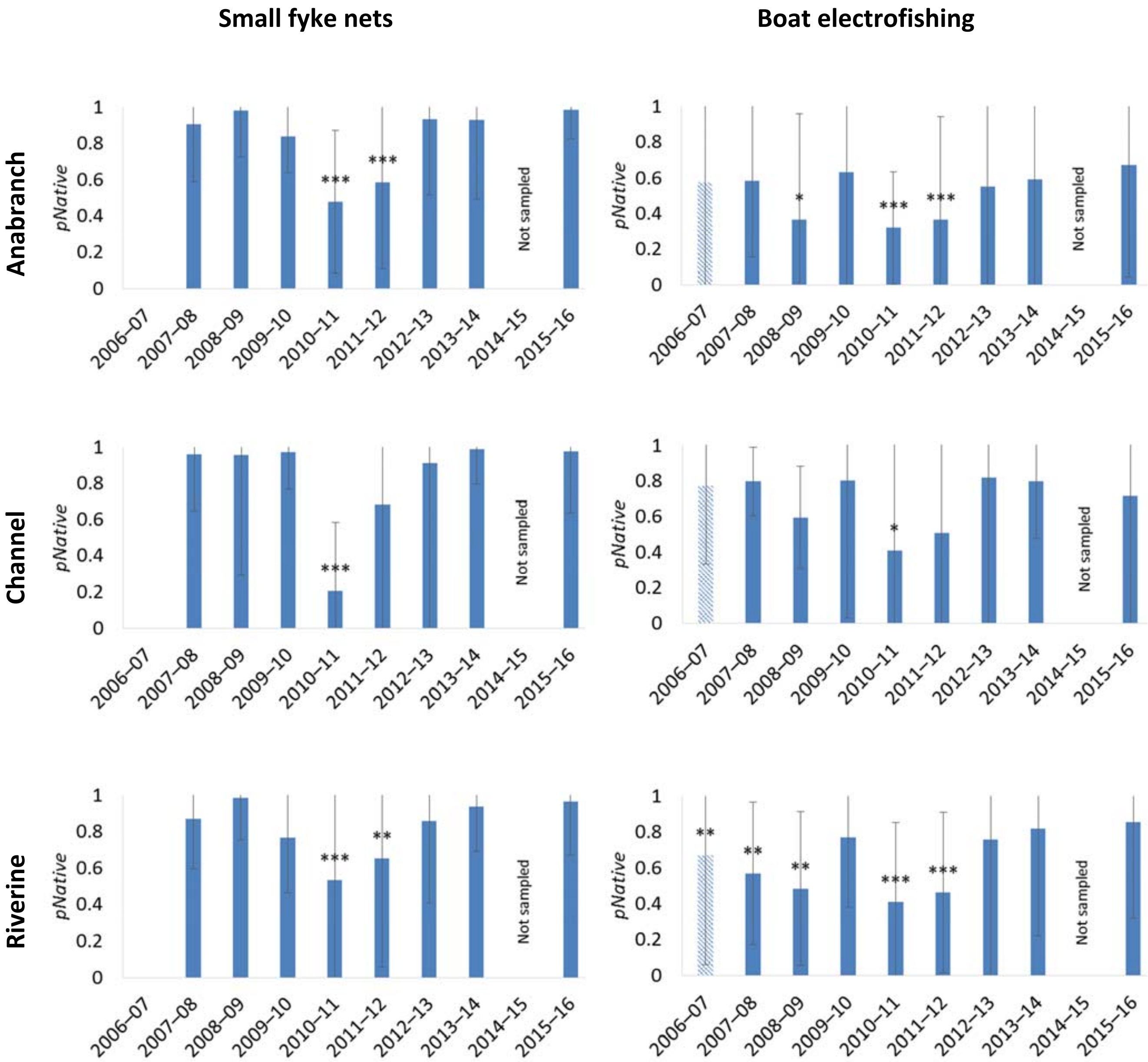


Figure 8-5. Mean proportion (± SE) of total fish biomass as native fish biomass (*pNative*) for each year sampled in each macrohabitat with small fyke nets (left column) and with boat electrofishing (right column); ***, ** or * denote a significant difference between the sample year and the 2015–16 survey year at $P < 0.001$, $P < 0.01$ and $P < 0.05$, respectively.

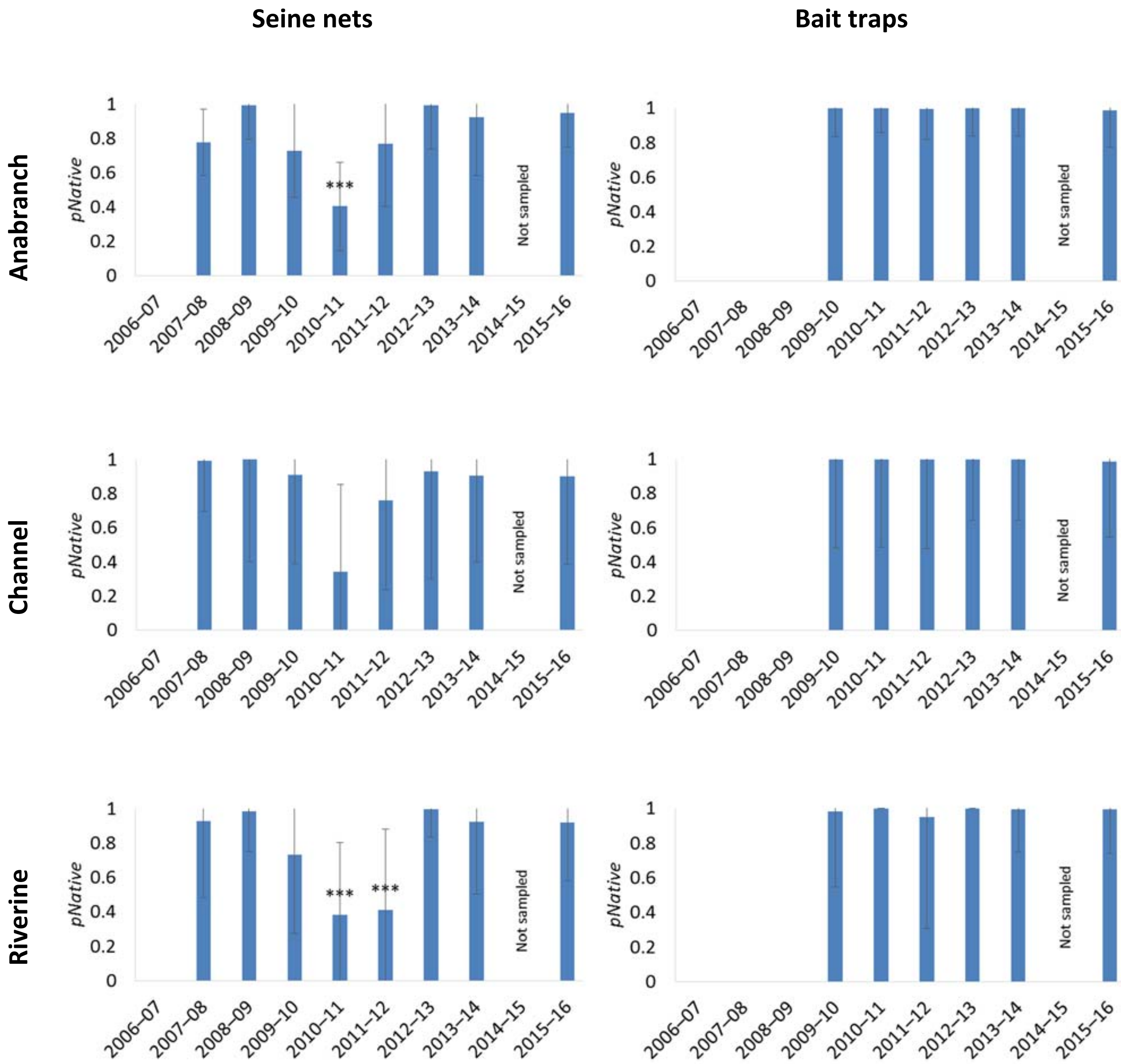


Figure 8-6. Mean proportion (± SE) of total fish biomass as native fish biomass (*pNative*) for each year sampled in each macrohabitat with seine nets (left column) and bait traps (right column); *** denotes a significant difference between the sample year and the 2015–16 survey year at $P < 0.001$.

8.4.5 Extent

Pairwise comparisons of 95% confidence intervals show no significant differences in extent index scores between years for any of the macrohabitats (Figure 8-7). That is, with respect to the extent index there was no clear difference between years in the distribution of species across macrohabitats. Notably, 95% CI were quite large, suggesting there is a low level of sensitivity associated with this index.

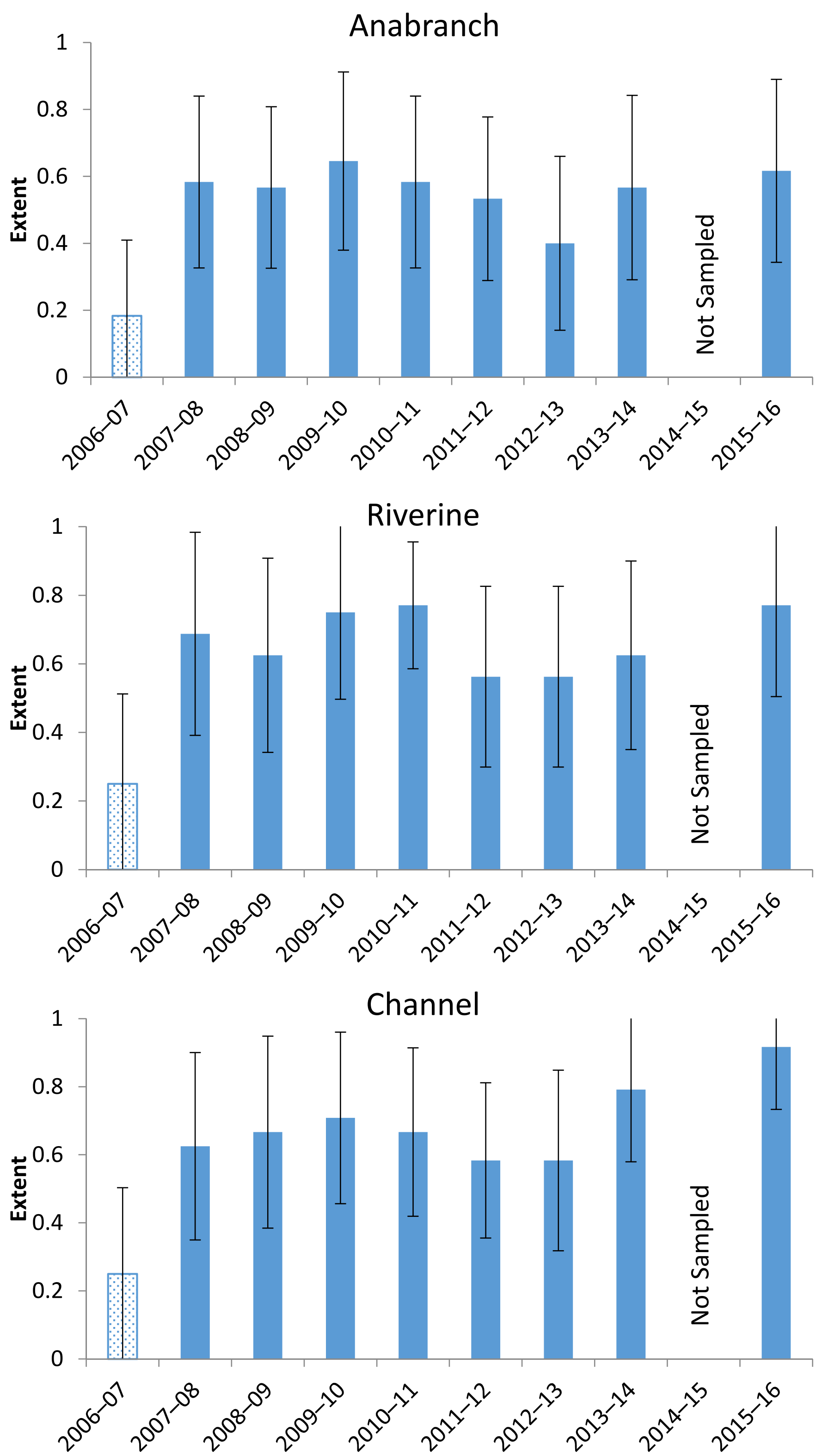


Figure 8-7. Mean (± CI) extent of native species observed in each macrohabitat, where a maximum score of 1 indicates all extant species were recorded at all sites within the macrohabitat.

Note: small fyke nets were included in the sampling design from 2007–08 (i.e. not used in 2006–07; dotted bars).

8.5 Discussion

The Living Murray condition monitoring for fish at LMW has spanned four years of drought (2006–07 to 2009–10), a flood year (2010–11), and four post-flood years (2011–12 to 2015–16; 2014–15 not sampled). The response of the fish community to the hydrological variation over this period has occurred primarily with respect to changes in the abundance of Freshwater catfish, the appearance of Spangled perch, and the spatial fragmentation of many small-bodied species following the 2010–11 flood and blackwater event (Henderson et al. 2013). However, the metrics that were introduced in 2013–14 for analysis of fish monitoring data are quite variable with respect to their sensitivity to fish community dynamics at LMW. Where there were significant or notable differences they were generally macrohabitat-specific. However, the metrics employed to date present a pattern of resilience in the icon site native fish community consistent with the overall objectives of relative abundance and diversity (MDBA 2012b).

Murray cod numbers sampled during 2015–16 are the highest since before the flood year (2008–09) and show signs of a successful spawning season and recruitment from previous years. Figure 8-1 (left) demonstrates a typical length–frequency histogram with large young-of-year (YOY) and juvenile cohorts compared to adult life-history stages. The lower numbers of Murray cod between 400–700 mm total length sampled may relate to the legal recreational fishing slot limit of 550–750 mm.

Golden perch numbers sampled during 2015–16 are the highest since during the flood and subsequent high-flow monitoring years (2010–11 and 2011–12). The length–frequency histogram for Golden perch demonstrates a population dominated by adults, with few YOY fish recruiting to juveniles (Figure 8-1 left). Golden perch are classified as flood-spawners and so it is not expected that they will spawn in years of low or no flow; nor show annual signs of a strong recruitment response.

Alpha diversity for the 2015–16 monitoring year was the highest across all macrohabitats with low variance compared to most other years (Figure 8-2). This is primarily due to the majority of species being recorded in each site at each reach. Variation exists where some of the lesser-recorded species (i.e. Flathead gudgeons, Spangled and Silver perch) are absent. Flathead and Dwarf flathead gudgeons and Silver perch often display patchy distribution and generally occur in lower abundance while Spangled perch are opportunistic flood-dispersers, hence their association with flood and post-flood years (Ellis et al. 2015). Apart from these less abundant species, α-diversity results indicate that the adopted objective "*Maintain native fish populations, their relative abundance and diversity",* is being met.

Similarly, β-diversity (Figure 8-3) was influenced primarily by the presence or absence of less-frequently encountered species such as Spangled perch, Silver perch, Freshwater catfish and Flathead gudgeons. Apart from these species, β-diversity is relatively consistent across years with minimal changes in the presence or distribution of species across all macrohabitats. These results indicate that the adopted objective is being met.

Species expectedness was reached or exceeded across all macrohabitats during the 2015–16 sampling season (Figure 8-4). There has been an increase in diversity towards and exceeding the expected species value of 6.7 since 2012–13 across all macrohabitats. For the anabranch macrohabitat 2015–16 was the first occurrence of average species diversity reaching the expected value. Because the expectedness metric produces an expected species value that is comparable to α-diversity for each macrohabitat it is recommended that this metric continue to be used as an indication of diversity in place of a separate α-diversity index. Both α-diversity and expectedness results can be inferred from the one metric.

Native fish biomass as a proportion of total fish biomass was significantly elevated in the present survey year, compared to the minima of 2010–11 and 2011–12, for most gear types and most macrohabitats (Figure 8-5 and Figure 8-6). In comparison with the two previous survey years, this year's data show little change since the recovery of native fish biomass observed in 2012–13, suggesting that biomass-domination by nonnatives following the high flows of 2010 and 2011 was brief then quickly reversed and stabilised in subsequent years. This resilience in native fish biomass suggests that non-native species such as Eastern Gambusia and Common carp have not recruited strongly in the icon site from the abundant juveniles observed in 2010 and 2011, relative to the proliferation of native fish observed in recent years.

For the extent metric there is a large overlap of 95% confidence intervals (± CI) between years, suggesting there is a low level of sensitivity associated with this index (Figure 8-7). While there is a general trend of increase in extent since 2012–13 across all macrohabitats, it is not significant.

8.5.1 Program design refinement considerations

Linear mixed modelling is appropriate for ongoing analyses of the whole dataset where an unbalanced design results from missing values occurring due to sites omitted during drier times. For future monitoring, increased power to detect change would result from increasing the number of reaches sampled.

Bait traps, small fyke nets and seine nets are sampling similar 'edge' habitats and appear to have some similar catch characteristics. An examination of the continued utility of all three gears is warranted. Small fyke nets are set 'overnight' and to lift them requires a second visit to each site. Significant economies in sampling may be possible if they could be omitted from the suite of gears without compromising the 'richness' of the data gathered. Bait traps also seem to catch little that has not already been sampled adequately by small fyke nets and/or seine nets. While time in the field to set and retrieve bait traps is not onerous, there are also minor costs associated with data processing analysis and reporting. An examination of bait-trap data utility would be worthwhile.

References

Allen DM, Service SK, Ogburn-Matthews MV (1992) Factors Influencing the Collection Efficiency of Estuarine Fishes. *Transactions of the American Fisheries Society* **121**, 234-244.

ANCA (1996) A Directory of Important Wetlands in Australia, p. 157. Australian Nature Conservation Agency, Canberra.

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

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

Baldwin D (1999) Dissolved organic matter and phosphorus leached from fresh and 'terrestially' aged river red gums: implications for assessing river-floodplain interactions. *Freshwater Biology* **41**, 675-685.

Bates D, Maechler M, Bolker B, Walker S (2014) lme4:Linear-mixed effects models using Eigen and s4. R package version 1.1-6. In. Available from: http://CRAN.R-project.org/package=lme4. Accessed:

Bayes E, Cook D, Jolly K, Robertson P (2010) Lindsay-Wallpolla frog and aquatic vegetation surveys 2009-2010, p. 153. Report prepared for the Mallee Catchment Management Authority as part of Murray–Darling Basin Authority funded project MD 1458 by Australian Ecosystems Pty Ltd, Patterson Lakes, VIC.

BOM (2011) Climate Data Online. In. Commonwealth of Australia. Available from: http://www.bom.gov.au/index.shtml. Accessed: 25/10/2011

BOM (2014) Climate Data Online. In. Bureau of Meteorology. Available from: http://www.bom.gov.au/index.shtml. Accessed: 08/05/2014

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

Boulton A, Brock M, Robson B*, et al.* (2014) *Australian Freshwater Ecology: Processes and Management*, Second edn. John Wiley & Sons, Ltd, West Sussex, UK.

Briggs S, Maher M (1983) Litter fall and leaf decomposition in a River Red Gum (*Eucalyptus camaldulensis*). *Australian Journal of Botany* **31**, 307–316.

Briggs SV, Lawler WG, Thornton SA (1997) Relationships Between Hydrological Control of River Red Gum Wetlands and Waterbird Breeding. *Emu* **97**, 31–42.

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

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

Brown P, Henderson M, Freestone F, Campbell C (2015) The Living Murray condition monitoring refinement project report updates for the Lindsay–Mulcra–Wallpolla Islands and Hattah Lakes icon sites 2014–15. Final report prepared for the Mallee Catchment Management Authority by the Murray–Darling Freshwater Research Centre, MDFRC Publication 84/2015, June, 34 pp.

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

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

Chen H, Zamorano MF, Ivanoff D (2013) Effect of deep flooding on nutrients and non-structural carbohydrates of mature Typha domingensis and its post-flooding recovery. *Ecological Engineering* **53**, 267–274.

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

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

Ecological Associates (2007) Floodplain options investigation: Lindsay, Mulcra and Wallpolla Islands. Final Report prepared for the Mallee Catchment Management Authority by Ecological Associates. 122 p.

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

Environment Australia (2001) A directory of important wetlands in Australia, p. 157. Environment Australia, Canberra, ACT.

Flanery F, Clark E (2007) Cumbungi - *friend or foe?* You asked for it. Hot topics in native vegetation management. Greening Australia.

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

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

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

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

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

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

Hayes D, Ferreri C, Taylor W (1996) Active Fish Capture Methods. In: *Fisheries Techniques, 2nd edition* (eds. Murphy B, Willis D), pp. 193-220. American Fisheries Society, Bethesda, Maryland.

Henderson M (2010) River Red Gum. In: *The Living Murray Condition Monitoring at Lindsay, Mulcra and Wallpolla Islands 2009/10* (ed. Henderson M), pp. 7–55. Report prepared for the Department of Sustainability and Environment by The Murray–Darling Freshwater Research Centre, Mildura, VIC.

Henderson M, Freestone F, Wood D*, et al.* (2013) The Living Murray Condition Monitoring at Lindsay, Mulcra and Wallpolla Islands 2012–13: Part A – Main Report. Final Report prepared for the Mallee Catchment Management Authority by The Murray–Darling Freshwater Research Centre. MDFRC Publication 18/2013, 153 p.

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

Hubert W (1996) Passive Capture Techniques. In: *Fisheries Techniques, 2nd edition* (eds. Murphy B, Willis D), pp. 157–181. American Fisheries Society, Bethesda, Maryland.

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

Johns C, Campbell C (2011) Assessing the importance of water-borne propagule movement for establishment of aquatic vegetation in a long-dry wetland. Final report prepared for the Murray–Darling Basin Authority by The Murray–Darling Freshwater Research Centre, MDFRC Publication 13/2011. Mildura, VIC. 58 p.

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

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

Kolz AL (2006) Electrical Conductivity as Applied to Electrofishing. *Transactions of the American Fisheries Society* **135**, 509–518.

Laarman PW, Ryckman JR (1982) Relative Size Selectivity of Trap Nets for Eight Species of Fish. *North American Journal of Fisheries Management* **2**, 33–37.

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

Lyons J (1986) Capture Efficiency of a Beach Seine for Seven Freshwater Fishes in a North-Temperate Lake. *North American Journal of Fisheries Management* **6**, 288–289.

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

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

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

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

MDBA (2010) Fish Condition Monitoring Approach (v.2), p. 20. Murray–Darling Basin Authority, Canberra.

MDBA (2012a) Ground-based survey methods for The Living Murray assessment of condition of river red gum and black box populations. Murray–Darling Basin Authority. Canberra, ACT. 58 p.

MDBA (2012b) Lindsay-Wallpolla Islands: Environmental Water Management Plan 2012, p. 47. Murray–Darling Basin Authority, Canberra, ACT.

MDBC (2002) Native Fish Strategy. Murray–Darling Basin Commission. Canberra, ACT.

MDBC (2003) Preliminary investigations into observed river red gum decline along the River Murray below Euston. Murray–Darling Basin Commission. Canberra, ACT. 36 p.

MDBC (2005) The Living Murray Foundation Report on the significant ecological assets targeted in the First Step Decision. Murray–Darling Basin Commission. Canberra, ACT. 324 p.

MDBC (2006) The Chowilla Floodplain and Lindsay-Wallpolla Islands Icon Site Environmental Management Plan 2006-2007. Murray–Darling Basin Commission. Canberra, ACT. 160 p.

MDBC (2007a) The Living Murray Outcomes Evaluation Framework. Murray–Darling Basin Commission. Canberra, ACT. 31 p.

MDBC (2007b) Sustainable Rivers Audit Protocols – Approved Manual for Implementation Period 4: 2007–08. Murray–Darling Basin Commission, Canberra.

MDBMC (2003) Murray–Darling Basin Ministerial Council Communiqué. 7 p.

MDFRC (2011) The Living Murray: Condition Monitoring Program design for Chowilla Floodplain and the Lindsay, Mulcra and Wallpolla Islands. A report prepared for the Murray–Darling Basin Commission by The Murray–Darling Freshwater Research Centre. Mildura. 154 p.

Murphy B, Willis D (1996) *Fisheries Techniques, 2nd edition*, 2nd edn. Published by American Fisheries Society, Bethesda, Maryland.

Nichols SJ, Robinson WA, Norris R (2006) Sample variability influences on the precision of predictive bioassessment. In: *Evaluation and Application of Methods for Biological Assessment of Streams* (eds. Norris RH, Milligan A), pp. 215–233. Springer.

Nicol J, Weedon J (2006) Understorey vegetation monitoring of the Chowilla River Red Gum watering trials. Aquatic Sciences, South Australian Research and Development Institute. RD04/0177, Adelaide, SA. 83 p.

Overton I, McEwan K, Gabrovsek C, Sherrah J (2006) The River Murray Floodplain Inundation Model (RiM-FIM): Hume Dam to Lower Lakes. CSIRO. Canberra, ACT. 140 p.

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

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

Reynolds J (1996) Electrofishing. In: *Fisheries Techniques, 2nd edition* (eds. Murphy B, Willis D), pp. 221–253. American Fisheries Society, Bethesda, Maryland.

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

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

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

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

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

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

Sainty G, Jacobs S (1981) *Waterplants of New South Wales.* Water Resources Commission, NSW.

Scholz O, Reid J, Wallace T, Meredith S (2007a) The Living Murray Initiative: Lindsay-Mulcra-Wallpolla Islands and Hattah Lakes Icon Sites Condition Monitoring program design. Report to the Mallee Catchment Management Authority. Report No. 1/2007. The Murray–Darling Freshwater Research Centre. Mildura. 63 p.

Scholz O, Sharpe C, Fraser P, Henderson M, Ellis I (2007b) The Living Murray Initiative: Lindsay-Mulcra-Wallpolla Islands and Hattah Lakes Icon Sites 2006-7 condition monitoring program data. Report to the Mallee Catchment Management Authority. MDFRC LBL Report No. 2/2007. The Murray–Darling Freshwater Research Centre.

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

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

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

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