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Commonwealth Environmental Water Office Long-Term Intervention Monitoring Project 2014–2019:
Lower Murray River
Technical Report

A report prepared for the Commonwealth Environmental Water Office by the South Australian Research and Development Institute, Aquatic Sciences

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Disclaimer

TABLE OF CONTENTS

TABLE OF CONTENTS I

LIST OF FIGURES II

LIST OF TABLES VII

ACKNOWLEDGEMENTS IX

EXECUTIVE SUMMARY 1

1 INTRODUCTION 4

1.1 Flow regimes and riverine ecology 4
1.2 CEWO Long-Term Intervention Monitoring Project 4
1.3 Expected outcomes in the LMR 6
1.4 Environmental water delivery 7
1.5 Purpose of the CEWO LTIM report for 2014–2019 14

2 INDICATORS 15

2.1 Hydrological Regime 15
2.2 Stream Metabolism 30
2.3 Matter Transport 48
2.4 Microinvertebrates 64
2.5 Fish Spawning and Recruitment 88
2.6 Fish Assemblage 104

3 SYNTHESIS AND EVALUATION 116

4 GENERAL MANAGEMENT RECOMMENDATIONS 121

5 REFERENCES 123

6 APPENDICES 133

APPENDIX A: EXPECTED OUTCOMES OF COMMONWEALTH ENVIRONMENTAL WATER IN THE LMR, LOWER LAKES AND COORONG 133

APPENDIX B: OVERVIEW OF OTHER WATERING AND MANAGEMENT ACTIVITIES FROM 2014–2019 134

APPENDIX C: DEW EVALUATION QUESTIONS 145

APPENDIX D: SUPPLEMENTARY INFORMATION FOR INDICATORS 157

ACRONYMS 165

GLOSSARY 166

LIST OF FIGURES

Figure 1. Map of the LMR Selected Area showing the floodplain (blue), gorge (green) and swamplands (orange) geomorphic zones, and the Lower Lakes, Coorong and Murray Mouth (yellow). Sampling sites are indicated by coloured circles. Fish Spawning and Recruitment sites represent larval sampling only. 5

Figure 2. The various flow types of the LMR as described by the Murray–Darling Basin Plan and Gawne *et al.* (2013). 6

Figure 3. Cause and effect diagram of flow for the main channel of the LMR with respect to the proposed indicators. Magnitude, timing and duration are factors of flow (in black). Yellow indicators followed standard protocols to support quantitative Basin-wide and Selected Area evaluation, where applicable. Purple indicators were developed to address objectives and test a series of Selected Area-specific hypotheses with respect to biological/ecological response to environmental flows. 7

Figure 4. Daily flow (ML/d) in the LMR at the South Australian border (blue solid line) from January 1996 to July 2019, compared to modelled flow under natural conditions (grey dashed line). Approximate bankfull flow in the main channel of the LMR is shown (black dashed line). 9

Figure 5. Flow to South Australia from July 2014 to July 2019 showing the contribution of environmental water. CEW = Commonwealth environmental water. Other eWater = The Living Murray, Victorian Environmental Water Holder, New South Wales Department of Planning, Industry and Environment and water delivered as part of River Murray Increased Flows. Specific watering events (Table 3) are shown, except base flows and weir pool manipulations. The 'no eWater' component includes South Australian entitlement held by the Commonwealth Environmental Water Holder and by TLM. Approximate bankfull flow (red dotted line) is 45,000 ML/d in the LMR. Modelled flows under natural conditions (dotted black line) peaked at 163,230 ML/d in 2016-17. 10

Figure 6. Source of all (environmental and consumptive) water delivered to the South Australian border (MDBA) from July 2014 to June 2019. Bigmod salinity routines used as a proxy for transport of biological matter, to estimate the proportion of the flow that originated at different upstream tributaries.Refer to Figure 31 for location of rivers and tributaries, relative to the LMR. 13

Figure 7. Discharge (Flow to South Australia), inundated area and length of river with faster flowing velocities (v>0.2 m/s and v>0.3 m/s) for the LMR between Locks 1 & 6 (excluding anabranches). Total length of river assessed in the LMR = 345 km.. 18

Figure 8. Median modelled velocity in each weir pool (line), with the range in velocities within the weir pool (the shaded area), defined by the 10th and 90th percentiles, in the LMR. 19

Figure 9. Modelled water level at the upstream and downstream ends of Weir Pool 5 in the LMR. 20

Figure 10. Modelled water level at the upstream end of each weir pool in the LMR. 21

Figure 11. Area inundated (hectares) in each weir pool for changes in discharge and weir pool level in the LMR. Weir Pool 3 is shown on a different colour scale due to the larger areas in this weir pool. 28

Figure 12. Proportion of the weir pool with velocities exceeding 0.3 m/s for changes in discharge and weir pool level in the LMR. 29

Figure 13. Rates of GPP (🔵), ER (🟠) and NEP (●) at (left) Lock 6 and (right) Lock 1 over the five monitoring periods from 2014–2019. 34

Figure 14. The response of GPP per unit phytoplankton biomass in carbon units, to the mean irradiance of the water column over the daylight period for Lock 6 (orange circles) and Lock 1 (blue circles) when water velocities were greater than 0.22 m/s. Regression, y = 0.0164x - 0.0024, r^2= 0.78. 35

Figure 15. (A) Environmental water contributions to discharge at Lock 6, (B) Fractional change in volumetric GPP(b), (C) Fractional change in cross-sectional GPP(b), due to flow including all environmental water (🔵) or only CEW (🟠).................. 36

Figure 16. Flow induced changes at Lock 6 (top) and Hattah (bottom) in (left) cumulative volumetric GPP over each monitoring period, (right) cumulative cross-sectional production over each monitoring period, due to the addition of CEW (blue bars) and all environmental water (grey) from no environmental water. Fractions above each bar represent proportional changes in GPP due to environmental flows compared with no environmental flows (e.g. 1.02 indicates GPP increased by 2% over no environmental water)... 37

Figure 17. Comparison of the vertical attenuation coefficient kd measured in the field and calculated from the dissolved organic carbon concentration and the turbidity.
38

Figure 18. The vertical attenuation coefficient for Photosynthetically Active Radiation (●) and the contributions due to DOC (🔵) and NTU (🟠) at Lock 6 for each monitoring period. 39

Figure 19. Respiration rates and net production for phytoplankton (●), bacteria (🟡) and both combined (🟠). 40

Figure 20. (A) The gas exchange coefficient at different water velocities and (B) the water velocities with (🔵) and without (🟠) environmental water at Lock 6. 42

Figure 21. Overview of model domain applied in the Matter Transport study of LTIM using TUFLOW-FV. Grid provided courtesy of DEW. Coloured grids in maps on the right-hand side represent depths, i.e. increasing depth from shallow (blue) to deep (red). 51

Figure 22. Overview of the three flow scenarios assessed by the model simulations. Scenarios include flow with all water, flow without Commonwealth environmental water (no CEW) and flow without any environmental water (no eWater). Flows were applied to the matter transport model at the upstream boundary, which is at Lock 1.. 52

Figure 23. Modelled cells (circled) used for calculating the modelled concentration of nutrients or salt at the Wellington, Lake Alexandrina Middle and Murray Mouth sites.. 54

Figure 24. Modelled daily salinity concentrations (left) and six monthly cumulative salt exports (net, right) with and without environmental water delivery for 2013–2019. Scenarios include with all water, without Commonwealth environmental water (no CEW) and without any environmental water (no eWater)........................... 56

Figure 25. Mean microinvertebrate density collected in the LMR at below Lock 6 in the floodplain geomorphic zone, demonstrating community assemblage characteristics in a) 2014-15, (b) 2015-16, (c) 2016-17 and (d) 2017-18, plotted against flow discharge (blue line) and temperature at SA Border (red line). ... 73

Figure 26. Mean microinvertebrate density collected in the LMR at below Lock 1 in the floodplain geomorphic zone, demonstrating community assemblage characteristics in a) 2014-15, (b) 2015-16, (c) 2016-17 and (d) 2017-18, plotted against flow discharge (blue line) and temperature at SA Border (red line). ... 74

Figure 27. Mean (±S.E.) species richness of microinvertebrates collected in the LMR at sites below Lock 6 (black bars) and Lock 1 (grey bars). 75

Figure 28. MDS ordination of microinvertebrate assemblage data (square-root transformed) at Lock 6 and Lock 1 during 2014–2018. Samples are labelled by sampling trip. Species density correlations are overlaid with blue vectors and water temperature (Water temp) and discharge into South Australia (QSA) correlations are overlain with red vectors. Species have been labelled by their family and preferred habitat (FamilyHabitat). Rotifer labels are blue and microcrustacean (cladocerans and copepods) labels are in black. Correlation value set to 0.5... 76

Figure 29. Abundance (number, *n*) of prey identified in the guts of post-flexion Murray cod (*n* = 15; TL = 7.8–12.0 mm) and golden perch (*n* = 13; TL = 6.5–20 mm) from 2014–2018. Prey taxa are presented as copepods (black), cladocerans (blue), rotifers (red), malacostracans (green), insects (grey) and oligochaetes (purple). * indicates littoral microcrustaceans. 80

Figure 30. Abundance (number, *n*) of prey identified in the guts of post-flexion freshwater catfish (*n* = 20; TL = 13.0–20.7 mm) from 2014–2018. Prey taxa are presented as copepods (black), cladocerans (blue), ostracods (orange), malacostracans (green) and insects (grey). * indicates littoral microcrustaceans....................... 81

Figure 31. Map showing the location of the Murray–Darling Basin and the major rivers that comprise the southern Murray–Darling Basin, the numbered Locks (L) and Weirs (up to Lock 26, Torrumbarry), the Darling, Lachlan, Murrumbidgee, Edward–Wakool, Campaspe and Goulburn rivers and Lake Victoria, an off-stream storage used to regulate flows in the Murray River. 89

Figure 32. (a) Mean ^{87}Sr/^{86}Sr (with minimum and maximum values as error bars) in water samples collected from spring–summer in the mid-Murray (Barmah, Torrumbarry and Lock 11), lower River Murray (Lock 9, 6 and 1) and Darling rivers from 2011–2019, and (b) annual discharge (GL) in the Murray River at the South Australian

border (QSA) and the proportion of discharge from the Darling River at Burtundy that contributed to QSA. 93

Figure 33. Mean catch-per-unit-effort (CPUE, individuals per trip) ± standard error of (a) golden perch larvae and (b) fish eggs collected below Lock 6 and Lock 1, in the LMR, from 2010-11 to 2018-19 during larval tow sampling. From 2014-15 onwards, natal origins of larvae (expressed in text as proportions) are inferred through a subsample of otolith core $^{87}Sr/^{86}Sr$ from larval and golden perch of that spawn year. LRM = Murray River, below the Darling confluence. Perch (i.e. silver or golden perch) eggs were not differentiated from other eggs prior to 2015-16. Daily flow to South Australia (SA) (solid blue line), modelled flow under non-regulated (natural) conditions (dotted grey line) and approximate bankfull flow (dotted black line) are shown. 95

Figure 34. Back-calculated spawn dates (frequency) for larval golden perch captured at Lock 6 (blue bars) and Lock 1 (red bars) in the LMR during spring/summer 2014–2019. Spawn dates are plotted against discharge (ML/d) at the South Australian border (solid blue line) and downstream of Lock 1 (solid red line), and water temperature (°C) (dotted black line). 96

Figure 35. Mean catch-per-unit-effort (CPUE) ± standard error of golden perch captured during Category 1 Fish Assemblage electrofishing (individuals per 90 second shot) in the gorge geomorphic zone (10 sites) of the LMR in autumn from 2015–2019. CPUE data from five sites are presented for 2017 as other sites were sampled during winter 2017. 97

Figure 36. Age frequency distribution of golden perch from the LMR from 2015–2019 showing the natal origins of dominant cohorts inferred from otolith core signatures of the sampled fish in comparison to the water sample reference collection (Figure 32). LRM = Murray River, below the Darling confluence. Percentage of origin for each cohort are based on the subsampled population. Age cohorts with grey bars were not assessed for natal origin. 99

Figure 37. An individual life history profile based on transect analysis of $^{87}Sr/^{86}Sr$ from the core to edge of an otolith from a (a) 2016-17 year class (age 1+) golden perch from Lowbank, (b) 2010-11 year class (age 8+) golden perch from Cobdogla, (c) 2005-06 year class (age 11+) golden perch from Rilli Island and (d) 1996-97 year class (age 21+) golden perch from Swan Reach. Green dashed line indicates the temporally stable water $^{87}Sr/^{86}Sr$ of the lower Darling River (i.e. ~0.7075) and the blue dashed lines represent the range of water $^{87}Sr/^{86}Sr$ in the lower River Murray (i.e. ~0.7080–0.7160). Red dashed lines represent the range of water $^{87}Sr/^{86}Sr$ in the mid-Murray River (Lock 11–Torrumbarry, ~0.7160–0.7190) (Figure 32). 100

Figure 38. Target species for the LMR: (a) Murray cod and (b) freshwater catfish (equilibrium life history); (c) golden perch and (d) silver perch (periodic life history); and (e) carp gudgeon, (f) Murray rainbowfish and (g) bony herring (opportunistic life history). 104

Figure 39. Mean catch-per-unit-effort (CPUE) ± standard error of (a) large-bodied fish species captured using electrofishing (individuals per 90 second shot) and (b)

small-bodied fish species captured using fine-mesh fyke nets (individuals per net per hour) in the gorge geomorphic zone (10 sites) of the LMR in autumn from 2015–2019. Electrofishing CPUE data from five sites are presented for 2017 as other sites were sampled during winter 2017. 106

Figure 40. Non-metric multi-dimensional scaling (MDS) plot of (a) large-bodied fish assemblages sampled by electrofishing and (b) small-bodied fish assemblages sampled by fyke netting in the gorge geomorphic zone of the LMR from 2015–2018. Sites (*n* = 5) sampled in winter 2017 were removed from the ordination. 108

Figure 41. Length frequency distributions and age structures of golden perch collected from the gorge geomorphic zone of the LMR from 2015–2019. 109

Figure 42. Length frequency distributions and age structures of silver perch collected from the gorge geomorphic zone of the LMR from 2015–2019. 110

Figure 43. Length frequency distributions and age structures of freshwater catfish collected from the gorge geomorphic zone of the LMR from 2015–2019. 111

Figure 44. Length frequency distributions and age structures of Murray cod collected from the gorge geomorphic zone of the LMR from 2015–2019. 112

Figure 45. Length frequency distributions and age structures of bony herring collected from the gorge geomorphic zone of the LMR from 2016–2019. 113

LIST OF TABLES

Table 1. Total annual volumes (gigalitres) of environmental water (eWater), including Commonwealth environmental water (CEW), delivered to the LMR (excludes wetland watering) and the proportion contribution towards total flow to the LMR (QSA). Volumes are provided by the CEWO and include the environmental components of the South Australian entitlement. Note that there are differences among data sources depending on whether water delivery by the end of a water year is based on accounted flows or flows physically delivered in real time. TLM = The Living Murray, VEWH = Victorian Environmental Water Holder, RMIF = River Murray Increased Flows, NSW DPIE = New South Wales Department of Planning, Industry and Environment. 8

Table 2. Annual flow over the Murray barrages (total volume, GL) from 2014–2019, showing contribution by Commonwealth environmental water (CEW). CEW and total flow volumes are based on SA barrage dashboard accounting data. Matter transport results in Section 2.3 are based on different modelled data, and may not necessarily reflect the accounted data presented here. 9

Table 3. Specific CEW watering events and expected outcomes for the LMR and Coorong, Lower Lakes and Murray Mouth from 2014–2019 (source, CEWO). Refer to Figure 5 for timing of events. 11

Table 4. Hydrological Regime evaluation questions and answers. CEW = Commonwealth environmental water, eWater = environmental water. 22

Table 5. Number of days in each monitoring period meeting the velocity criteria (0.18 m/s) without environmental water (eWater) contributions, with Commonwealth environmental water (CEW), or with all environmental water. 42

Table 6. Stream Metabolism evaluation questions and answers relating to Commonwealth environmental water (CEW) and environmental water (eWater). The Lock 6 site has been used to answer the evaluation questions. 43

Table 7. Water quality sampling sites within each water-body. 49

Table 8. Five year record of modelled salt export (tonnes) over the barrages to the Coorong estuary and through the Murray Mouth into the Southern Ocean. ... 55

Table 9. Five year record of modelled phytoplankton export (as carbon, tonnes) over the barrages to the Coorong estuary........ 57

Table 10. Matter Transport evaluation questions and answers. CEW = Commonwealth environmental water. 58

Table 11. Microinvertebrate sampling dates from 2014–2018 in the LMR. 65

Table 12. Details of microinvertebrate sampling sites downstream (DS) of Lock 1 and Lock 6 in the LMR. 66

Table 13. Sample sizes of large-bodied fish larvae analysed for gut contents from 2014–2018, indicating the number of fish with contents in their stomachs. Total lengths (TL) are for all fish sampled. 67

Table 14. PERMANOVA table of results for pairwise comparisons on density data between years, where 2014 = 2014-15, 2015 = 2015-16, 2016 = 2016-17 and 2017 = 2017-18, and sites where Lk1 = Lock 1 and Lk 6 = Lock 6. *P*-values presented in bold are significant comparisons. 68

Table 15. PERMANOVA table of results for pairwise comparisons on species richness data between years, where 2014 = 2014-15, 2015 = 2015-16, 2016 = 2016-17 and 2017 = 2017-18, and sites where Lk1 = Lock 1 and Lk 6 = Lock 6. *P*-values presented in bold are significant comparisons. 69

Table 16. PERMANOVA table of results for pairwise comparisons on community assemblage data between years, where 2014 = 2014-15, 2015 = 2015-16, 2016 = 2016-17 and 2017 = 2017-18, and sites where Lk1 = Lock 1 and Lk 6 = Lock 6. *P*-values presented in bold are significant comparisons. 69

Table 17. Table summarising the timing and origin of environmental water delivered to the LMR in 2014-15, 2015-16 and 2017-18 and the species indicated by SIMPER analysis driving major differences in the microinvertebrate community due to higher densities at the time of delivery and likey community origin. The sampling period 2016-17 is not included due to the very low propotion of environmental water delivered during that period. 77

Table 18. Microinvertebrates evaluation questions and answers. CEW = Commonwealth environmental water, eWater = environmental water. 82

Table 19. Locations of water sample collection for $^{87}Sr/^{86}Sr$ analysis from 2014–2019. 89

Table 20. Larval fish sampling details from 2014–2019. 90

Table 21. PERMANOVA pairwise comparison test results for golden perch electrofishing abundance (individuals per 90 second shot) in the gorge geomorphic zone (10 sites) of the LMR in autumn from 2015–2019. *P*-values presented in bold are significant comparisons, using Bonferroni corrected α = 0.0050 (Narum 2006) for comparisons between years (ten comparisons). 97

Table 22. Fish spawning and recruitment evaluation questions and answers. CEW = Commonwealth environmental water, YOY = young-of-year. 101

Table 23. PERMANOVA pairwise comparison test results for large- and small-bodied fish assemblages in the gorge geomorphic zone of the LMR from autumn 2015–2019. *P*-values presented in bold are significant comparisons, using Bonferroni corrected α = 0.0050 (Narum 2006) for comparisons between years (ten comparisons) 107

Table 24. CEWO evaluation questions by indicators for the Lower Murray River (LMR) and the Coorong, Lower Lakes and Murray Mouth (CLLMM). Evaluation questions are sourced or adapted from Gawne *et al.* (2014). Evaluation of CEW for hydraulic and matter transport questions is based on modelled data. CEW = Commonwealth environmental water. Refer to the evaluation in respective indicator sections (Section 2) for more detail. 120

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EXECUTIVE SUMMARY

From 2014-15 to 2018-19, the Commonwealth Environmental Water Office (CEWO) Long-Term Intervention Monitoring (LTIM) Project was implemented to monitor and evaluate ecological outcomes of environmental water delivery in the Lower Murray River (LMR), along with six other Selected Areas in the Murray–Darling Basin (MDB). The overall aim of this project was to demonstrate the ecological outcomes of Commonwealth environmental water delivery and support adaptive management.

The five-year project period was dominated by low flow conditions (<18,000 ML/d at the South Australian border) except for 2016-17, when there was unregulated, overbank flow (peak ~94,600 ML/d in December). Over the five years, Commonwealth environmental water (~3,440 GL, 76% of total environmental water use) predominantly contributed to baseflows and freshes in the LMR, particularly as winter and spring–early summer flow pulses. Seven indicators were selected for LTIM in the LMR, which focused on the main channel and aimed to answer Basin and/or local (Selected Area) evaluation questions.

Key findings and ecological outcomes

Environmental water delivery contributed to some ecological improvements in the LMR Selected Area from 2014-15 to 2018-19:

- Longitudinal connectivity: Commonwealth environmental water increased annual flow discharge in the LMR in all years, including helping to meet the Basin-wide environmental watering target of >30% increase in flows in the Murray River (calculated at the SA border) in two of the five years.
- Lateral connectivity: Commonwealth environmental water increased connection to riparian zones with greater inundated areas in the four low-flow years (total 2,306 ha), primarily via weir pool raisings.
- Hydraulic variability: environmental water increased the duration and extent of ‘flowing water’ habitat (velocities >0.3 m/s), particularly during 2017-18 and increased water level variability, particularly in tailwaters. This hydraulic variability would have benefited native plants and animals adapted to a riverine environment and variable water levels.
- Increased water mixing (velocities >~0.2 m/s) and oxygen exchange at the surface from environmental water reduced the risk of low dissolved oxygen, particularly during spring–summer – the critical period that corresponds with high ecosystem respiration rates and is the primary reproductive season of many species.
- River primary production, which supports aquatic food webs (e.g. invertebrates and fish), increased slightly in response to the physical changes generated by the addition of environmental water as the influence on channel volume was constrained by generally stable weir pool levels in the regulated LMR.
- Environmental water increased the transport of nutrients and phytoplankton, which would likely have stimulated primary and secondary productivity in downstream ecosystems, providing potential benefits to food webs of the LMR, Lower Lakes, Coorong and Southern Ocean, adjacent to the Murray Mouth.
- Flows over the barrages to the Coorong were almost continuous throughout all years due to Commonwealth environmental water, maintaining connectivity

between the river and the Coorong estuary to support a functioning river system and species' life history processes (e.g. migration of diadromous fish).

- Commonwealth environmental water substantially increased salt export out of the Basin (modelled mean 0.2 million tonnes per year), reduced salt import into the Coorong (2.7 million tonnes per year), and reduced salinity levels in the Coorong, which was considered to be crucial for maintaining estuarine habitats, species diversity and ecosystem functions during this period.
- Increased microinvertebrate diversity and density coincided with increased flows (in some cases in conjunction with infrastructure operation) supported by Commonwealth environmental water. Increased flows transported microinvertebrates from upstream sources to the LMR, and led to an increased prevalence of littoral organisms. Increased variability and diversity of microinvertebrates may improve productivity and community resilience, which is important for aquatic food webs and may support diversity in higher trophic organisms (e.g. larval fish).
- Some spawning of golden perch coincided with Commonwealth environmental water delivery and occurred in the LMR each year. However, there was no evidence of successful 'recruitment', i.e. survival to juvenile stage, to contribute to the broader population. The golden perch population is currently dominated by a few distinct and ageing cohorts in the LMR.

Key learnings and management implications

- Hydrodynamic restoration is fundamental to reinstating riverine ecosystem function in the highly regulated LMR. Environmental water can be used to help reinstate key features of the natural hydrograph to support hydrodynamic and ecosystem restoration; for example, to reinstate in-channel spring–early summer flow pulses >20,000 ML/d.
- With existing volumes of environmental water and delivery constraints, during dry years, reaching and sustaining flows >20,000 ML/d in the LMR is largely reliant on coordinating flow deliveries across the southern connected Basin, including flows from tributaries (e.g. Goulburn, Murrumbidgee, Darling rivers). Under wetter scenarios, flows >20,000 ML/d may be achieved by delivering environmental water with unregulated flows.
- In the LMR, increasing flows to 20,000–45,000 ML/d significantly improves hydraulic conditions (e.g. increased velocity and water level variability). Infrastructure management, particularly weir pool lowering, could also be considered to complement flows to achieve hydraulic rehabilitation and the promotion of lotic conditions. To inform flow management and maximise ecological outcomes, however, we need to better understand the effect of specific aspects of flow (e.g. timing, magnitude and duration) on ecological processes and the hydraulic requirements of flow-dependant species. Evidence from this and allied investigations suggest that under current constraints and weir operating regimes, flow rates >20,000 ML/d are likely required to significantly influence golden perch spawning and recruitment in the LMR.
- Environmental water contributing to flows >45,000 ML/d (above bankfull level) will increase inundation area considerably along the LMR, supporting off-channel processes and floodplain biota (e.g. floodplain understorey vegetation and tree

health). Overbank flow is also an integral part of the natural flow regime in maintaining ecosystem health of floodplain rivers.

- Environmental water delivery that promotes longitudinal and lateral connectivity will enhance the productivity in the LMR, via increased carbon and nutrient inputs and matter transport, and facilitate the transport and dispersal of aquatic biota (e.g. microinvertebrates, fish larvae).
- To improve riverine productivity in the LMR, environmental water delivery in conjunction with more natural water level changes are desirable. Further research will be required to identify favourable water level regimes.
- Environmental flows are pivotal in maintaining barrage flows and end-of-system connectivity in the MDB, particularly during low flow periods, when there would otherwise be negligible water and matter exchange between the Lower Lakes and Coorong. Barrage flows play a critical role in salt export from and reducing salt import to the MDB, maintaining estuarine conditions in the Coorong to facilitate important life history processes of estuarine species and reduce the risk of Murray Mouth closure.
- The timing of environmental flow delivery should continue to align with ecological objectives and consider biological processes and life history requirements (e.g. reproductive season of flow-dependent species in spring/summer, spawning migration of diadromous fishes in winter/spring, or reducing salinities and maintaining water levels in the Coorong during summer/autumn).
- Flow management should consider the source of water (i.e. origin), when possible, which can influence water quality (e.g. turbidity, dissolved organic carbon, the amount and form of nutrients), ecological processes (e.g. primary/secondary productivity) and subsequent biological responses.
- Furthermore, maintaining flow integrity from its source (e.g. Darling River, Murray upstream or major tributaries) to the end of the Murray River system is important to support broad-scale ecological processes and outcomes (e.g. improved productivity, migration of diadromous species, enhanced spawning, larval fish dispersal and recruitment of flow-dependent species).

More specific management considerations are provided in Section 2, based on ecological outcomes and findings from indicators.

1 INTRODUCTION

1.1 Flow regimes and riverine ecology

River regulation and flow modification have severely impacted riverine ecosystems throughout the world, including the Murray–Darling Basin (MDB) (Maheshwari *et al.* 1995; Kingsford 2000). The southern MDB is highly regulated, where natural flow regimes have been substantially altered, leading to decreased hydrological (e.g. discharge) and hydraulic (e.g. water level and velocity) variability, and reduced floodplain inundation (Maheshwari *et al.* 1995; Bice *et al.* 2017). The Murray River downstream of the Darling River junction (herein, the lower River Murray) is modified by a series of low-level (<3 m) weirs (Figure 1), changing a connected flowing river to a series of weir pools (Walker 2006). The flow regime has been further exacerbated by upstream diversions and increased extraction. These have had profound impacts on riverine processes and ecosystems (Walker 1985; Walker and Thoms 1993; Wallace *et al.* 2014).

Natural flow regimes play a critical role in maintaining the distribution and abundance of native aquatic biota, and the ecological integrity of floodplain rivers (Junk *et al.* 1989; Poff *et al.* 1997; Bunn and Arthington 2002). Lotic (flowing water) habitats are important for ecological and life history processes for many native biota that are adapted to flowing riverine environments. For example, they provide stimuli for spawning of flow-cued species (e.g. golden perch *Macquaria ambigua*) (King *et al.* 2016), facilitate downstream drift and transportation of plankton, macroinvertebrates and fish larvae, and provide diverse hydraulic habitats that are suitable for a range of species (e.g. Murray cod, *Maccullochella peelii* and Murray crayfish, *Euastacus armatus*) (Mallen-Cooper and Zampatti 2018). Increased variability in water levels improve lateral connectivity and increase transport of material from off-channel habitats to enhance productivity and support food webs (Baldwin *et al.* 2016), benefit fringing and floodplain vegetation (e.g. Cooling *et al.* 2010), and assist in the regular "re-setting" of biofilms (Steinman and McIntire 1990), which are key components of riverine food webs.

Environmental flows have been used to re-establish key features of the natural flow regime for ecological restoration of the MDB (MDBA 2012; Koehn *et al.* 2014; Webb *et al.* 2017). The main channel of the Murray River, which includes the South Australian section (herein, Lower Murray River, LMR), represents a significant ecological asset to be targeted for environmental watering (MDBC 2006; DEWNR 2015). To achieve the greatest ecological benefits from available environmental water, it is important to understand biological and ecological responses to flow regimes. This provides critical knowledge to underpin environmental flow management in the LMR.

1.2 CEWO Long-Term Intervention Monitoring Project

In 2014, the five-year (2014-15–2018-19) Commonwealth Environmental Water Office (CEWO) Long-Term Intervention Monitoring (LTIM) Project was established to monitor and evaluate ecological outcomes of Commonwealth environmental water delivery in the MDB. The project was implemented across seven Selected Areas throughout the MDB, including the LMR, to assess and evaluate both Basin-scale and Selected Area (local) responses to environmental flows. The overall aim of this project was to demonstrate the ecological outcomes of Commonwealth environmental water delivery and support adaptive management.

In the LMR, the CEWO LTIM Project focused on the main river channel between the South Australian border and Wellington, with only one targeted investigation (i.e. Matter Transport) extending to the Lower Lakes and Coorong (Figure 1). The riverine monitoring sites (for indicators) covered three geomorphic zones (floodplain, gorge and swamplands) (Figure 1).

A total of seven indicators were established to assess ecological responses to environmental water delivery in the LMR. Three indicators (***Hydrology (channel)***[a], ***Stream Metabolism*** and ***Fish (channel)***) followed standard protocols to support quantitative Basin-wide and Selected Area evaluation, where applicable (Hale *et al.* 2014). Four indicators (***Hydrological Regime***, ***Matter Transport***, ***Microinvertebrates*** and ***Fish Spawning and Recruitment***) were developed to address Selected Area-specific objectives and test a series of hypotheses with respect to biological/ecological response to environmental flows.

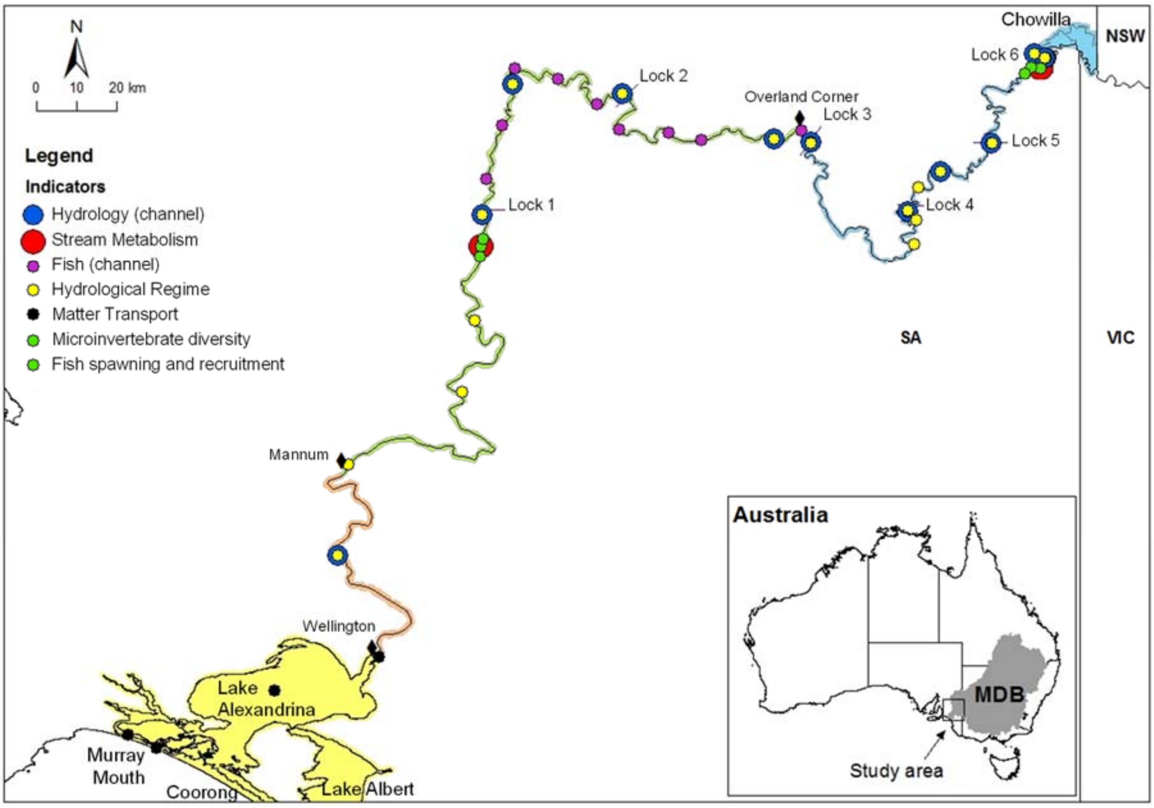

Figure 1. Map of the LMR Selected Area showing the floodplain (blue), gorge (green) and swamplands (orange) geomorphic zones, and the Lower Lakes, Coorong and Murray Mouth (yellow). Sampling sites are indicated by coloured circles. Fish Spawning and Recruitment sites represent larval sampling only.

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[a] Hydrology (Channel) does not directly address any specific CEWO evaluation question, but provides fundamental information for analysis and evaluation of monitoring outcomes against hydrological conditions and environmental water delivery for all other indicators. Results for this indicator are presented in Section 1.4.

1.3 Expected outcomes in the LMR

Within the five-year monitoring and evaluation period, Commonwealth environmental water in the LMR Selected Area was primarily used to contribute to baseflows and freshes in the Murray River channel (Figure 2), and to provide flows for the Lower Lakes and Coorong. These particular flows intended to achieve a variety of outcomes including those relating to fish, birds vegetation, river function, Lower Lakes water levels, salt export and connectivity between freshwater, estuarine and marine environment (Table 3; Appendix A), although only some of these were monitored through this project.

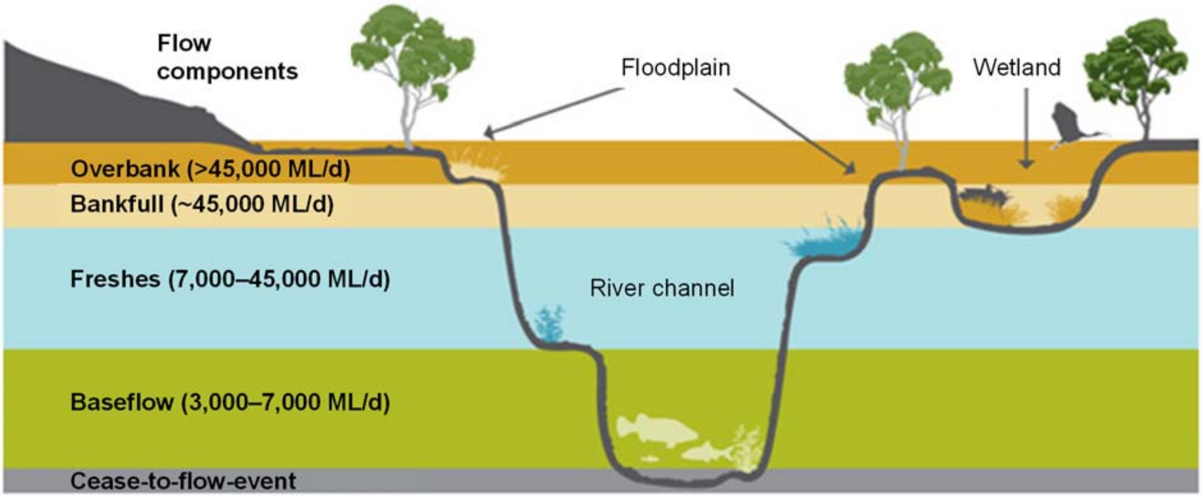

Figure 2. The various flow types of the LMR as described by the Murray–Darling Basin Plan and Gawne *et al.* (2013).

In the LMR, it was anticipated that environmental water delivery (contributing to baseflows and freshes) would: increase stream velocity, mixing and dilution; increase variability in water levels; and increase the inundated area of littoral zone of channels, low-lying wetlands and floodplains. These changes to hydrological/hydraulic conditions within the LMR were expected to lead to:

- Increased productivity due to lateral transport of organic material
- Increased transport of dissolved and particulate matter (salt and nutrients) downstream due to mobilisation and increased discharge
- Increased microinvertebrate diversity and abundance due to higher water residence times in littoral and off-channel habitats from increased water levels and discharge.
- Increased larval abundance of flow-dependent fish species due to the provision of flow-cues for spawning and increased larval drift and dispersion
- Increased recruitment of flow-dependent fish species due to increased spawning and larval drift, and enhanced survival rate due to increased productivity.

Over the long-term (decades), environmental water delivery is expected to make a significant contribution to achieving ecosystem outcomes in the LMR, through restoring ecological processes and improving habitat for biota in the main channel and floodplain/wetlands. A consolidated view of the expected outcomes driven by flow for the LMR is presented in Figure 3 below, which includes core monitoring indicators of the

LTIM Project. The conceptual diagram demonstrates the inter-relationships between the changes of hydrological/hydraulic regime and riverine productivity (stream metabolism, microinvertebrates) and matter transport, and how these may influence spawning and recruitment of flow-cued spawning fish and the overall fish assemblage in the LMR.

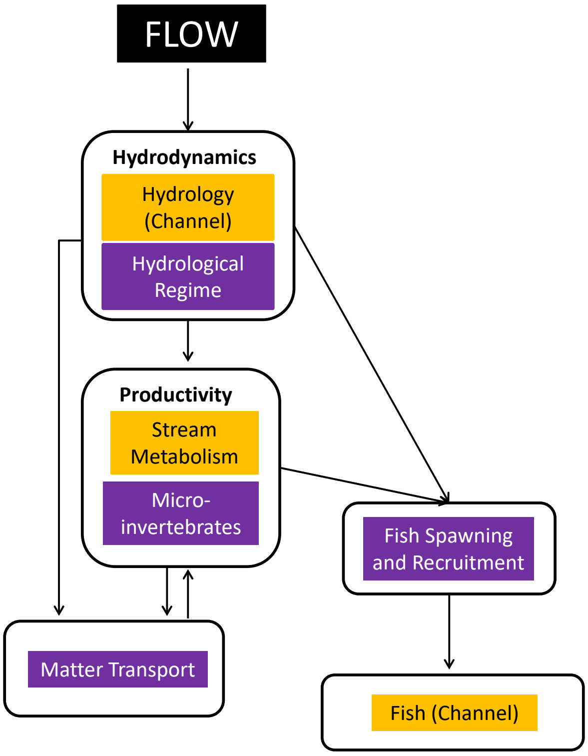

Figure 3. Cause and effect diagram of flow for the main channel of the LMR with respect to the proposed indicators. Magnitude, timing and duration are factors of flow (in black). Yellow indicators followed standard protocols to support quantitative Basin-wide and Selected Area evaluation, where applicable. Purple indicators were developed to address objectives and test a series of Selected Area-specific hypotheses with respect to biological/ecological response to environmental flows.

1.4 Environmental water delivery

Since 2011-12, environmental water has been delivered to the LMR (Table 1) to facilitate ecosystem recovery post ten years (2001–2010) of drought and restore ecological health (www.environment.gov.au/water/cewo). During the LTIM Project (2014-15 to 2018-19), a total of ~3,440 GL of Commonwealth environmental water was delivered to the LMR channel, in conjunction with other environmental flows (i.e. The Living Murray (TLM) Initiative, Victorian Environmental Water Holder, River Murray Increased Flows, and New South Wales Department of Planning, Industry and Environment). These deliveries largely occurred as return flows through coordinated watering events across the southern connected Basin to achieve multi-site environmental outcomes. Direct orders of environmental water to the South Australian border also occurred for specific purposes, often during summer–autumn, to provide flow for the Lakes and Coorong.

Table 1. Total annual volumes (gigalitres) of environmental water (eWater), including Commonwealth environmental water (CEW), delivered to the LMR (excludes wetland watering) and the proportion contribution towards total flow to the LMR (QSA). Volumes are provided by the CEWO and include the environmental components of the South Australian entitlement. Note that there are differences among data sources depending on whether water delivery by the end of a water year is based on accounted flows or flows physically delivered in real time. TLM = The Living Murray, VEWH = Victorian Environmental Water Holder, RMIF = River Murray Increased Flows, NSW DPIE = New South Wales Department of Planning, Industry and Environment.

Water year	2011-12	2012-13	2013-14	2014-15	2015-16	2016-17	2017-18	2018-19
CEW	329	786	576	581	798	618	894	549
TLM	139	289	107	107	101	234	176	51
VEWH			7	26	15	43	30	35
RMIF						100	53	111
NSW DPIE							9	
Total eWater	467 (5%)	1075 (15%)	690 (19%)	714 (25%)	914 (37%)	995 (11%)	1162 (43%)	747 (31%)

In four out of the last five years, which were hydrologically dry (i.e. flow <18,000 ML/d at the South Australian border, Figure 4), environmental water contributed to freshes in the LMR, particularly as winter and spring–early summer flow pulses (Figure 5), following upstream watering events. Winter flows (9,200 ML/d) in 2015 consisted of return flows from watering at Barmah–Millewa Forest and events in the Goulburn River, whilst winter (11,700 ML/d) and early spring flow pulses (8,700 ML/d) in the LMR during 2017 were attributed to return flows from the Goulburn and Murrumbidgee rivers (Figure 6). Winter flows (10,800 ML/d) in July 2018 were supported by return flows from the Goulburn River.

During spring 2014 and 2015, environmental water delivered in September/early October increased flow to ~10,000 ML/d and maintained flow at ~10,000–11,500 ML/d through October and November. In spring–early summer 2017, environmental water delivery supported multiple in-channel flow pulses in the LMR during mid-October (10,700 ML/d), early (17,800 ML/d) and late December (15,800 ML/d), contributed by flow pulses from the Goulburn River, and return flows from Barmah–Millewa Forest and Hattah Lakes (Figure 6). In-channel flow pulses in the LMR during late October 2018 (10,200 ML/d) and late December 2018 (12,100 ML/d) were supported by return flows from the Goulburn River and Barmah–Millewa Forest.

Unlike other years, there were high unregulated flows in 2016-17, resulting in overbank inundation (peak ~94,600 ML/d, Figure 4) during spring/early summer. The majority (~96%) of environmental flow was after mid-December 2016 (Figure 5). In this high flow year, environmental water originating from mixed upstream sources including the upper Murray (Hume), Goulburn–Broken system and the Darling River, assisted in slowing and extending the flood recession in summer (Figure 6).

During most LTIM years, environmental water delivery to the LMR from summer–late autumn was through direct trades at the South Australian border or return flows (e.g.

Goulburn and Darling River) (Figure 6). During this period, Commonwealth environmental water played a critical role in maintaining barrage releases (Table 2).

Furthermore, environmental water has been used for other complementary management actions to achieve ecological outcomes in the LMR (Appendix B). These included weir pool manipulations, operation of environmental regulators and wetland watering by pumping.

Key watering events and targeted outcomes for the LMR in the last five years are presented in Table 3. These events are also marked on Figure 5.

Table 2. Annual flow over the Murray barrages (total volume, GL) from 2014–2019, showing contribution by Commonwealth environmental water (CEW). CEW and total flow volumes are based on SA barrage dashboard accounting data. Matter transport results in Section 2.3 are based on different modelled data, and may not necessarily reflect the accounted data presented here.

Water year	2014-15	2015-16	2016-17	2017-18	2018-19
Total	987.1	561.0	6483.9	854.0	377.2
CEW	453.8	561.0	802.3	757.1	377.2

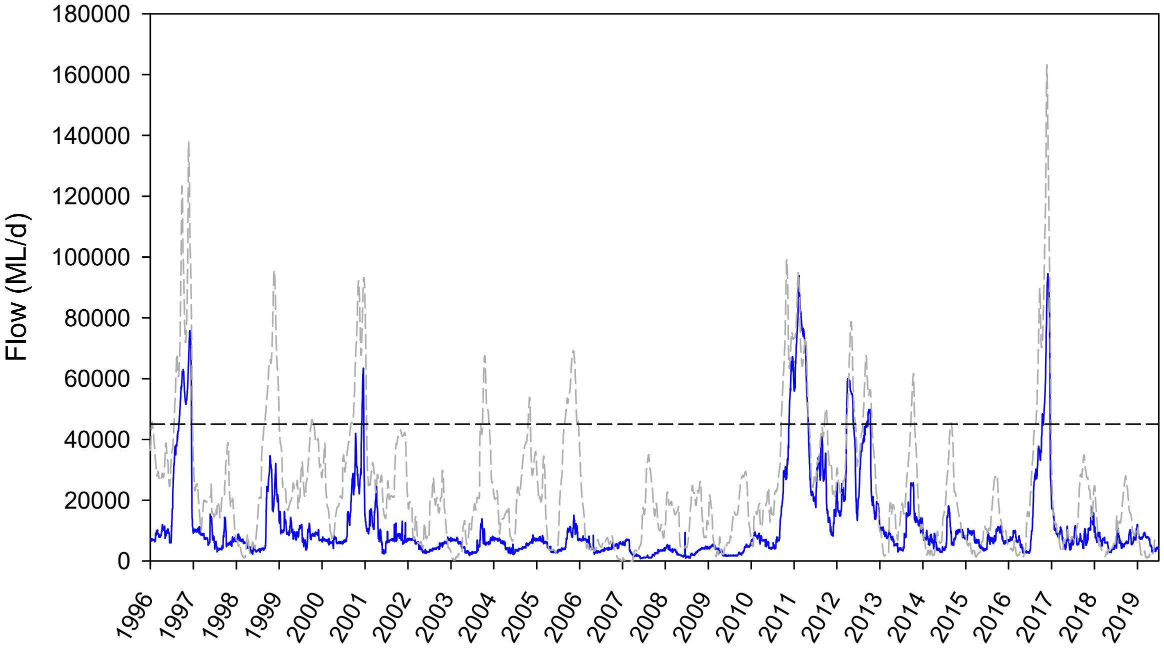

Figure 4. Daily flow (ML/d) in the LMR at the South Australian border (blue solid line) from January 1996 to July 2019, compared to modelled flow under natural conditions (grey dashed line). Approximate bankfull flow in the main channel of the LMR is shown (black dashed line).

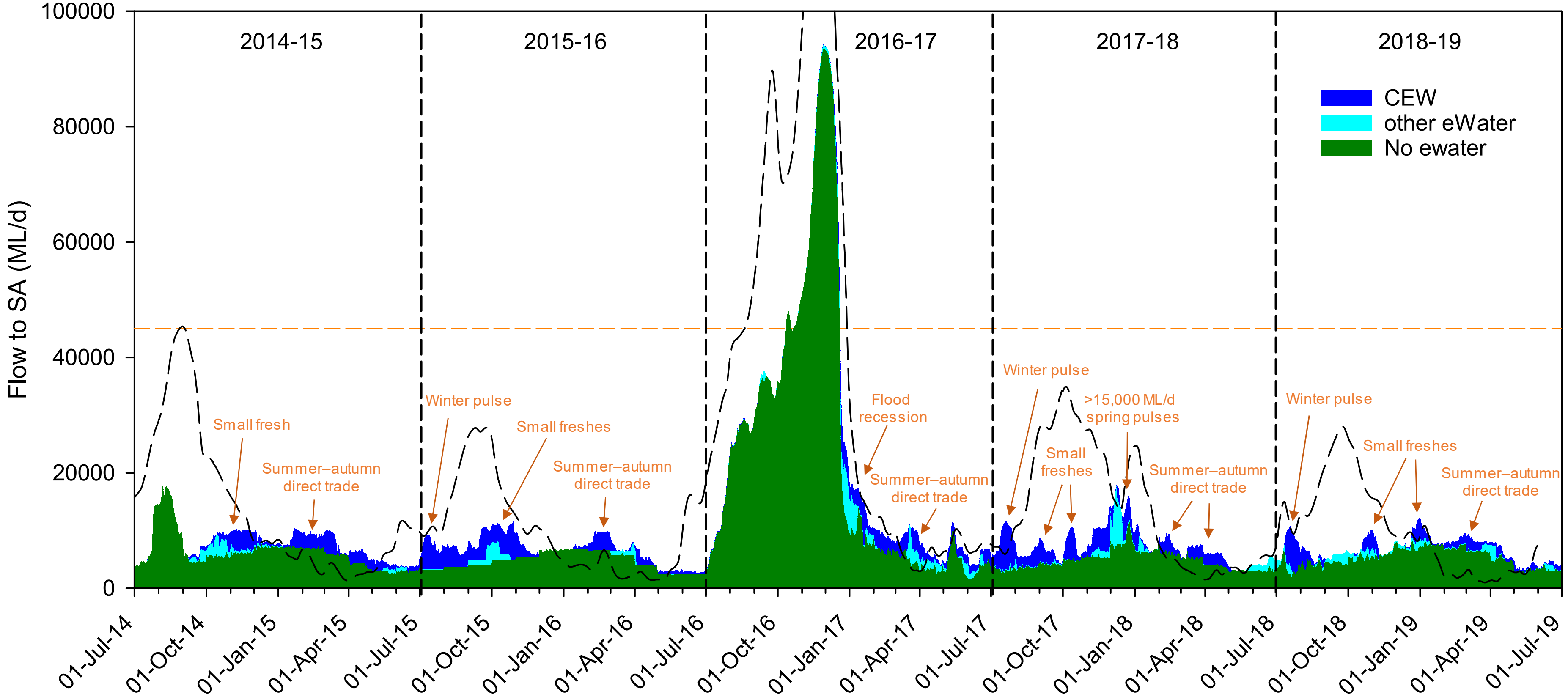

Figure 5. Flow to South Australia from July 2014 to July 2019 showing the contribution of environmental water. CEW = Commonwealth environmental water. Other eWater = The Living Murray, Victorian Environmental Water Holder, New South Wales Department of Planning, Industry and Environment and water delivered as part of River Murray Increased Flows. Specific watering events (Table 3) are shown, except base flows and weir pool manipulations. The 'no eWater' component includes South Australian entitlement held by the Commonwealth Environmental Water Holder and by TLM. Approximate bankfull flow (red dotted line) is 45,000 ML/d in the LMR. Modelled flows under natural conditions (dotted black line) peaked at 163,230 ML/d in 2016-17.

Table 3. Specific CEW watering events and expected outcomes for the LMR and Coorong, Lower Lakes and Murray Mouth from 2014–2019 (source, CEWO). Refer to Figure 5 for timing of events.

Flow event	Expected outcomes	Flow event delivered				
		2014-15	2015-16	2016-17	2017-18	2018-19
Base flows (<7,000 ML/d) and/or small freshes (7,000–15,000 ML/d)	• Fish diversity, distribution and breeding • Riparian and in-channel vegetation • Waterbird diversity, distribution and breeding • Riverine function • Coorong habitat • *Ruppia* recruitment and estuarine fish spawning • Lower Lakes salt export, water quality, water levels, vegetation and habitat provision Refer to Appendix A for more detail.	Y	Y	Y	Y	Y
Winter pulse (small, <15,000 ML/d)	• Fish habitat and condition • Riverine function • Freshening of Coorong • Diadromous fish migration • Coorong water quality/habitat suitability	N	9,200 ML/d	N	11,700 ML/d	10,800 ML/d
Spring–early summer pulse (>15,000 ML/d)	• Fish habitat and condition • Riverine function • Black bream spawning and recruitment • Coorong water quality/habitat suitability	N	N	N	17,800 & 15,800 ML/d	N
Summer–autumn direct trade	Elevated baseflows in the river: • Fish habitat and condition • Riverine function Supporting barrage releases in the Coorong: • Lower Lakes water levels • Freshening of Coorong	Y	Y	Y	Y	Y

Flow event	Expected outcomes	Flow event delivered				
		2014-15	2015-16	2016-17	2017-18	2018-19
	• Black bream spawning and recruitment (2017-18 & 2018-19) • Coorong water quality/habitat suitability • Vegetation diversity and migratory bird habitat fringing the Lower Lakes					
Flood recession flows	• Hypoxic blackwater mitigation	N	N	Y	N	N
Weir pool raising (R) and lowering (L)	Primary: • Lateral connectivity (R) • Food resources (R) • Vegetation (R) • Productivity (R) • Flowing habitat (L) • Drying of permanently inundated low-lying wetlands (L) Secondary: • Fish condition / spawning (R+L) • Waterbird habitat (R) • Salt and nutrient export (R+L)	N*	Y	N*	Y	Y

*Commonwealth environmental water underwrote the requirement of the weir-pools re-filling, which enabled the river operators to undertake the manipulations (See Appendix B).

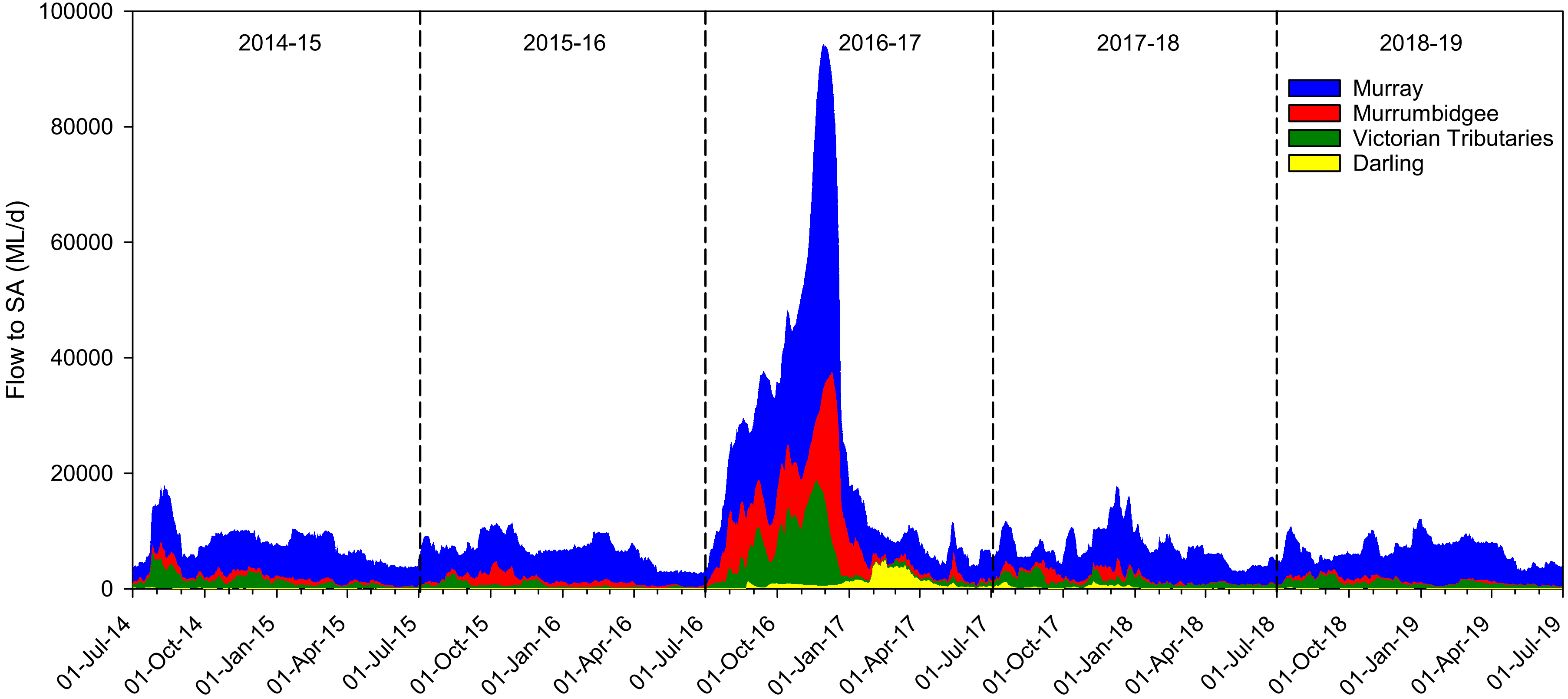

Figure 6. Source of all (environmental and consumptive) water delivered to the South Australian border (MDBA) from July 2014 to June 2019. Bigmod salinity routines used as a proxy for transport of biological matter, to estimate the proportion of the flow that originated at different upstream tributaries[b].Refer to Figure 31 for location of rivers and tributaries, relative to the LMR.

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[b] Molecules of water, nutrients, and the biological matter transported downstream often move slower than the wave front that is recorded as the change in flow discharge (Chow *et al.* 1988). To account for this, the MDBA has used Bigmod salinity routines as a proxy for transport of biological matter, to estimate the proportion of the flow at the South Australian border that originated at different upstream tributaries. While acknowledging potential difference in travel time between salt and other matter, this approach is preferred over estimating travel times based on observed changes in flow along the main channel.

1.5 Purpose of the CEWO LTIM report for 2014–2019

This report presents the key findings in the LMR over the five years of LTIM monitoring, and answers CEWO short-term (one-year) and long-term (five-year) evaluation questions (Sections 2 and 3). Refer to previous annual reports (Ye *et al.* 2016b; 2017; 2018; 2019) for more detailed evaluation of short-term responses to Commonwealth environmental water deliveries in the LMR, and the *Monitoring and Evaluation Plan for the LMR* (SARDI *et al.* 2018) for a more detailed description of methods. Specific management recommendations for environmental flows in the LMR are provided in Section 2, with general management implications summarised in Section 4, based on monitoring and evaluation outcomes, and expert knowledge. Monitoring and evaluation of Commonwealth environmental water delivery in the LMR from 2014-15 to 2018-19 focussed on spring–summer; therefore, our findings and recommendations on environmental water management are most relevant to this period.

2 INDICATORS

2.1 Hydrological Regime

Background

The discharge, or *hydrology*, in the LMR over the five-year period with environmental water was determined through routine monitoring. The hydrology expected to have occurred without environmental water components was determined by the Murray–Darling Basin Authority (MDBA) using a counterfactual modelling approach, as outlined in Section 1.4.

The hydraulic characteristics (e.g. depth or flow velocity) of fluvial ecosystems result from the interaction of discharge and physical features (e.g. channel morphology, woody debris, man-made structures, etc.), and have a profound influence on river ecosystem structure and function (Statzner and Higler 1986; Biggs *et al.* 2005; Bice *et al.* 2017) (also see Section 1.1). It is these hydraulic characteristics that biota can sense and respond to, i.e. a change in velocity or water level, rather than a change in discharge.

The purpose of this indicator was to quantify the changes in hydraulics due to the delivery of environmental water using hydraulic models, to provide a basis to infer ecological changes caused by environmental water. This approach is particularly important in the LMR where a given discharge may not produce the same hydraulic response, as downstream structures (weirs) will also influence the hydraulics occurring.

Hypothesis

Commonwealth environmental water will improve hydraulic metrics representing desirable riverine ecological conditions, for example increased velocities and increased variability in water levels.

Methods

A steady-state modelling approach was adopted, similar to that used in the Goulburn (Webb *et al.* 2015) and Edward-Wakool (Watts *et al.* 2015) Selected Areas. For each weir pool within the LMR Selected Area, i.e. Weir Pools 1 to 5, a range of steady state flow scenarios were simulated in the hydraulic models, between 2,000–100,000 ML/d, and a range of weir pool levels required to cover the range of conditions experienced during the five-year period. Models used for this analysis are outlined in McCullough *et al.* (2017) and Montazeri and Gibbs (2019). For each steady state scenario, a range of hydraulic metrics were computed, including the 10th, 50th and 90th percentile velocities within the weir pool, the proportion of the weir pool exceeding 0.2 and 0.3 metres per second (m/s), and water levels at regular locations along the weir pool. The earlier water velocity (>0.2 m/s) represents favourable velocities that entrain and transport/disperse phytoplankton, zooplankton and fish larvae (Gibbs *et al.* 2020) and allow gas exchange at the water surface (see Section 2.2 Stream Metabolism), and the latter (0.3 m/s) represents flowing water (lotic) conditions for riverine biota (Bice *et al.* 2017).

This steady-state approach also has the benefit of providing useful lookup information to inform future environmental water planning, for example the range in velocities present in a weir pool for a given discharge and weir pool level (see the following Management Implications section).

To enable a consistent comparison of in-channel velocity changes due to environmental water, the same area was used for all velocity analysis. The area used for velocity analysis for each weir pool comprised the inundated area at flows of 5,000 ML/d and normal pool level conditions. As changes to in-channel velocities were the focus of this analysis, this approach of eliminating additional areas inundated was considered reasonable. Additionally, where the full inundated area was used, the disproportionate increase in area of slow flowing backwaters compared to in-channel area as discharge increased had a large impact on the overall proportion of the weir pool with low velocities

Environmental water scenarios

With the lookup information derived from the hydraulic models, the time series of discharge for each of the environmental water scenarios presented in Section 1.4 and the downstream water level each day for each weir pool, time series of hydraulic parameters were interpolated using linear bivariate interpolation (R version 3.6.0 and akima package 0.6-2). Four scenarios have been considered based on the discharge data available, i.e. with all environmental water ("*AllWater*", representing observed conditions), without Commonwealth environmental water ("*NoCEW*"), without any environmental water ("*NoeWater*"), and a representation of natural conditions ("Without Development"). The discharge time series for these scenarios were provided by the MDBA at Locks 1, 3 and 5. Data for Locks 2 and 4 were interpolated using travel time information from the Source Murray Model. The discharge data within the LMR account for changes in diversions expected within South Australia with and without environmental water recovery by assuming full utilisation of the entitlements recovered for the environment in the without environmental water (*NoCEW* and *NoeWater*) scenarios.

The observed water levels were used as inputs for the *AllWater* scenario. For the without environmental water scenarios, the weir pool manipulations at Locks 2 and 5 (both lowering and raising, Appendix B) were removed in 2015-16, 2017-18 and 2018-19 because the manipulations were supported by environmental water, and instead the water level was assumed to be at normal pool level during these periods. Other weir pool raising events (for example Locks 1 and 2 in 2014-15 or Locks 2 and 5 in 2016-17, Appendix B) were expected to have occurred even if the environmental water was not available, and hence were maintained in the without environmental water scenarios.

Results

A summary of the results at the LMR Selected Area scale can be seen in Figure 7, presenting the discharge at the South Australian border for the different scenarios, the resulting area inundated, and length of the river between Lock 6 and Lock 1 experiencing lotic conditions, based on thresholds of velocity >0.2 m/s and >0.3 m/s.

Velocity

The modelling indicates that there were minor changes in the length of river with velocities exceeding 0.3 m/s due to environmental water from 1 July 2014 to 30 June 2019, with increases typically less than 25 km of the 345 km stretch of river (~7%) considered. The exception was 2017-18, where there was an increase in excess of 120 km of river (35% of the reach) with velocities greater than 0.3 m/s occurred for 6 days due to all environmental water (with a 60 km increase due to the Commonwealth environmental water contribution). This can be compared to the without development hydraulic conditions, where in all periods the whole section of river was expected to experience lotic conditions, even in relatively low flow years like 2015-16 and 2018-19. Similar patterns can be seen for the 0.2 m/s thresholds in Figure 7, with a number of events where 100 km (29%) of river exceeded this velocity due to environmental water over the five-year period, and an increase to 300 km (87%) of the river for a short period in 2017-18, with the majority of this due to environmental water (Figure 7).

The velocity magnitudes are presented on a weir pool basis in Figure 8, with the median velocity in the weir pool each day shown as a solid line, and the range in velocities (as the 10th and 90th percentiles) shown as the shaded band. The results suggest that that the largest increase in the reach with velocities exceeding 0.3 m/s created by environmental water occurred in Weir Pool 3, where the combination of a long weir pool (85 km) that has upper reaches less influenced by the downstream weir, and a shallower depth in this section of river, produced higher velocities without weir pool manipulation.

Water level and area

The inundation area expected for the different scenarios can be seen in the second panel of Figure 7, where flows, with the exception of 2016-17, were below bank full (approximately 45,000 ML/d), and hence changes in inundation due to environmental water were limited. The exceptions, seen as the increase in area for the AllWater scenario, are weir pool raising events in 2015-16, 2017-18 and 2018-19. The “without development” inundation areas in Figure 7 tend to be lower due to the influence of the weirs and locks maintaining higher water levels. This effect is demonstrated in Figure 9, where the upstream and downstream water levels for Weir Pool 5 for each scenario are presented. The upstream end of the weir pool is the least influenced by the downstream weir and hence most responsive to changes in discharge when the weirs are controlling water levels (below 54,000–67,000 ML/d, depending on the weir). The combination of environmental water and weir pool manipulation creates some variability in water levels at the upstream end of Weir Pool 5 that would not have occurred otherwise, and this variability tends to mimic the seasonality timing of without development pattern of river height, albeit at a smaller magnitude. At the downstream end of the weir pool, similar variability is seen in the without development scenario. However, the weir can be seen to maintain stable water levels in the without environmental water case, with the exception of the 2016-17 high flow event overtopping the weir (and a preceding weir pool raising). The weir pool raising events, and lowering in 2017-18, can also be seen in Figure 9. The upstream water levels for each weir pool along the LMR can be seen in Figure 10, which show similar patterns to Weir Pool 5.

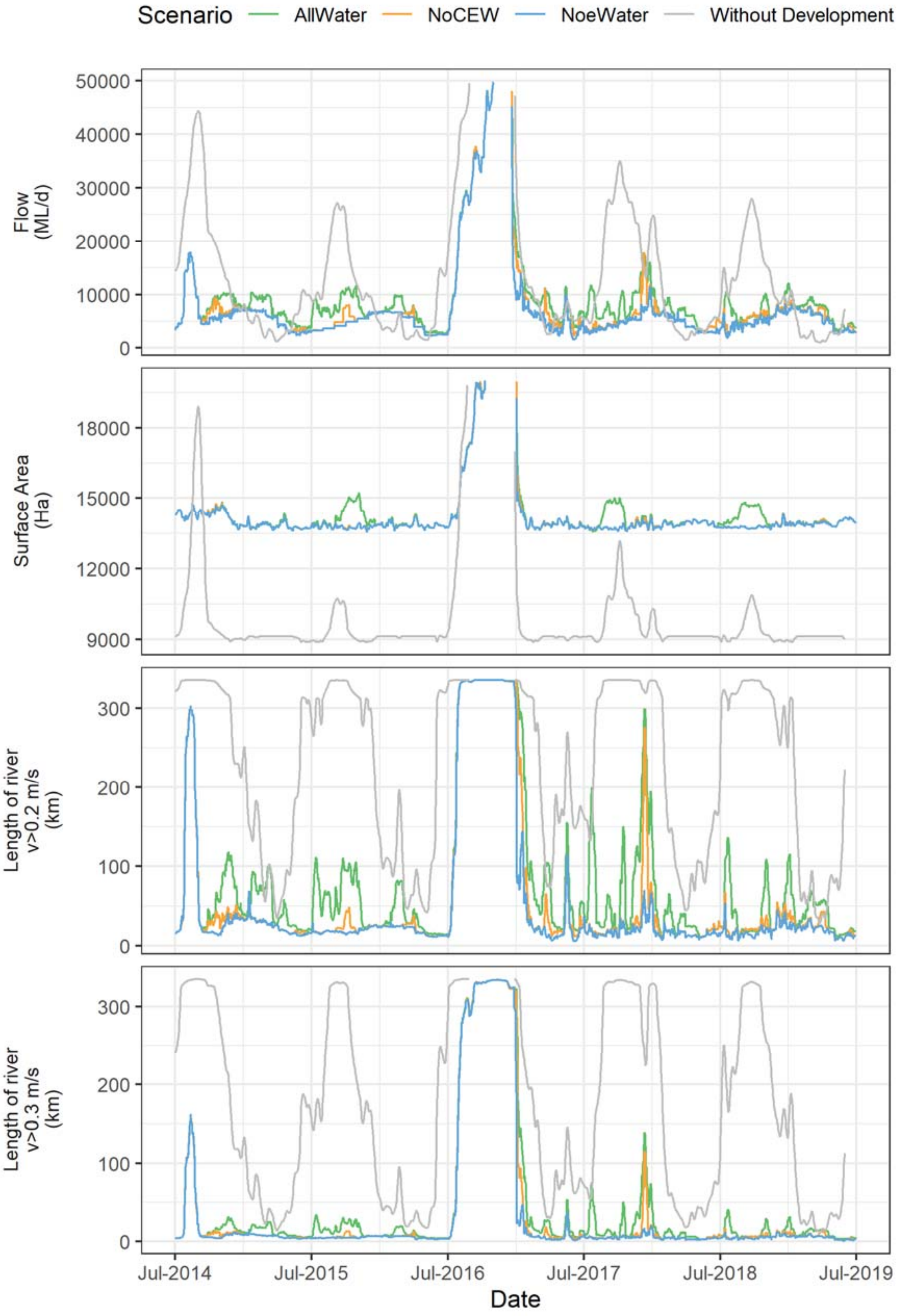

Figure 7. Discharge (Flow to South Australia), inundated area and length of river with faster flowing velocities (v>0.2 m/s and v>0.3 m/s) for the LMR between Locks 1 & 6 (excluding anabranches). Total length of river assessed in the LMR = 345 km.

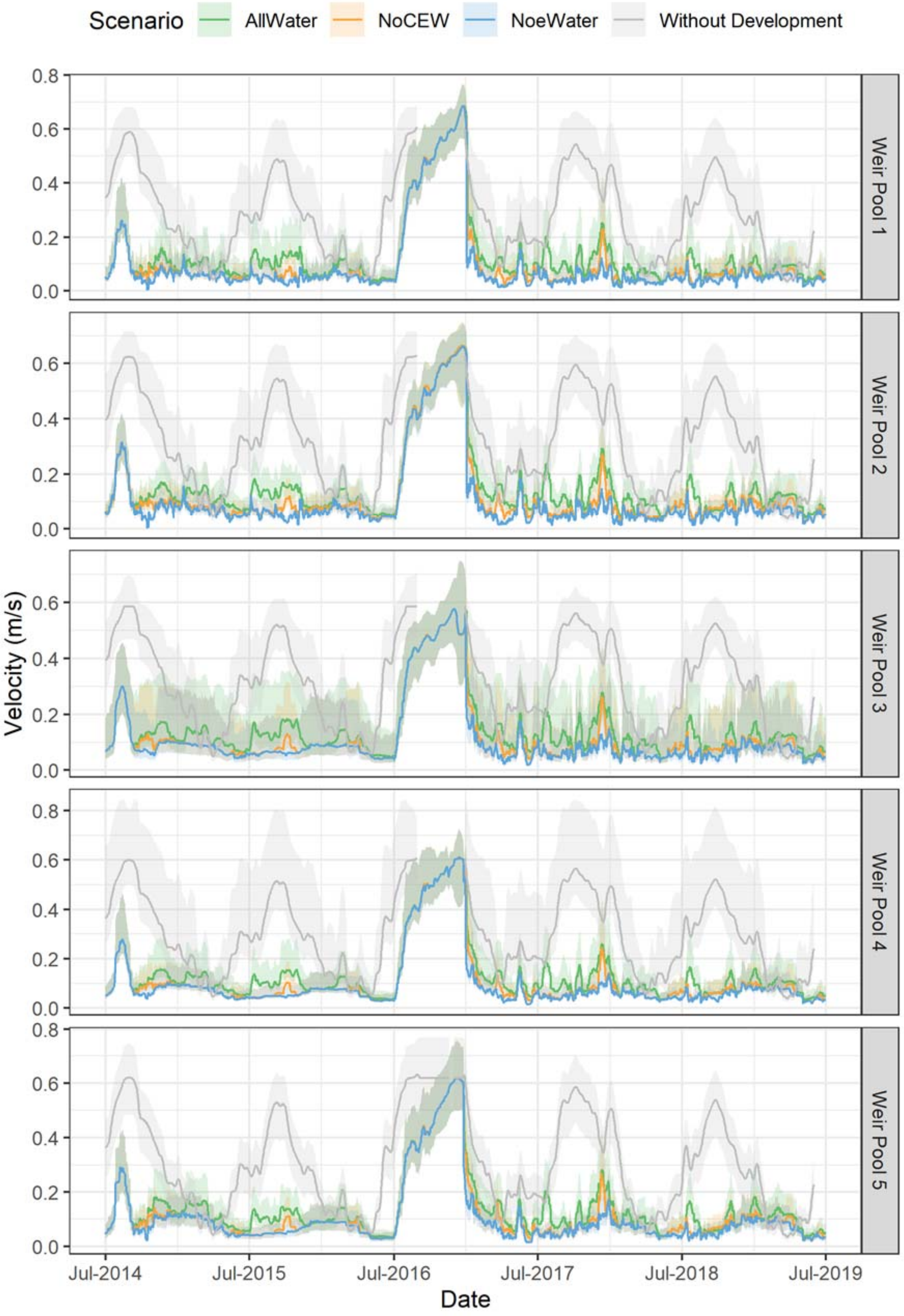

Figure 8. Median modelled velocity in each weir pool (line), with the range in velocities within the weir pool (the shaded area), defined by the 10th and 90th percentiles, in the LMR.

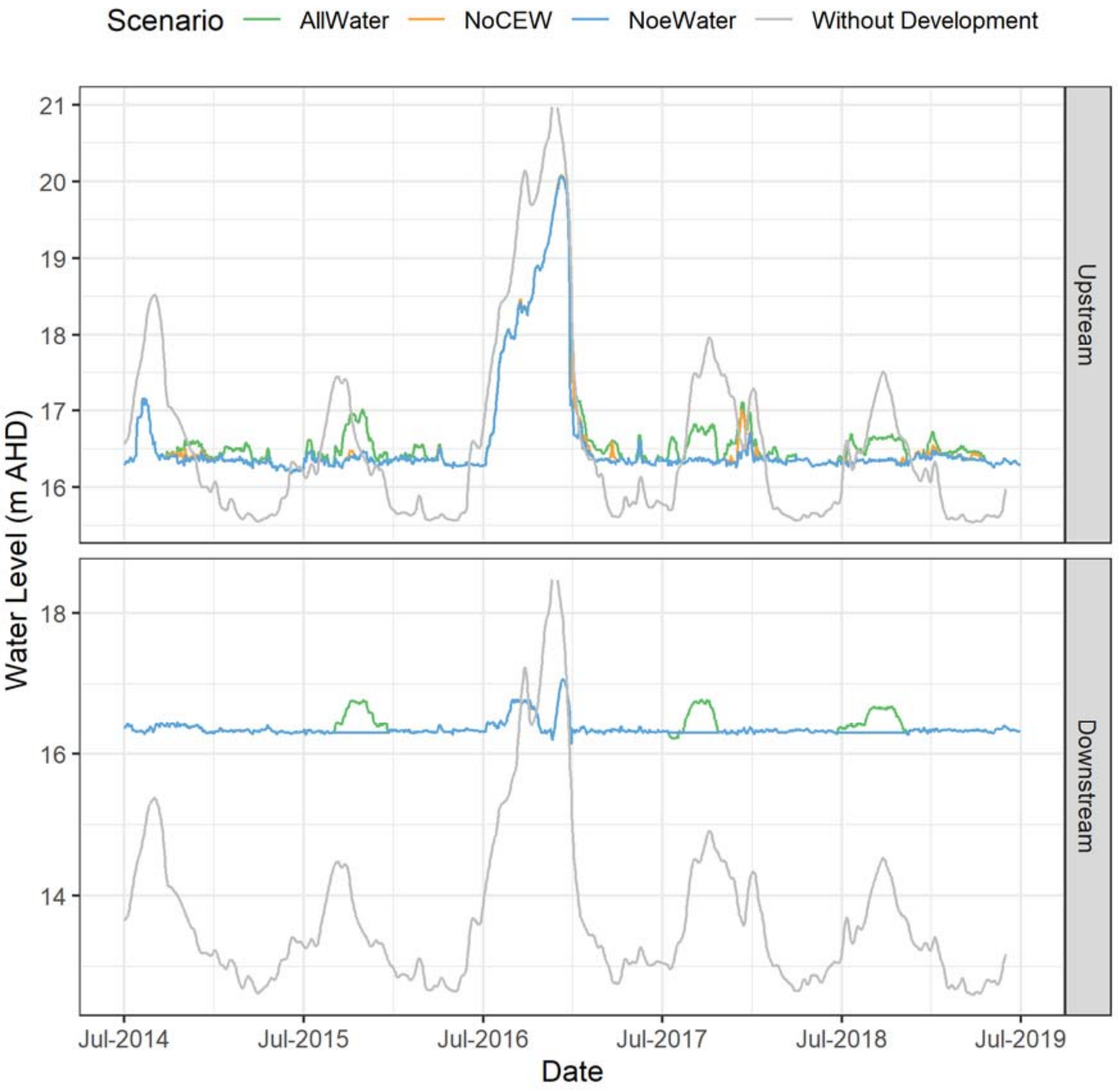

Figure 9. Modelled water level at the upstream and downstream ends of Weir Pool 5 in the LMR.

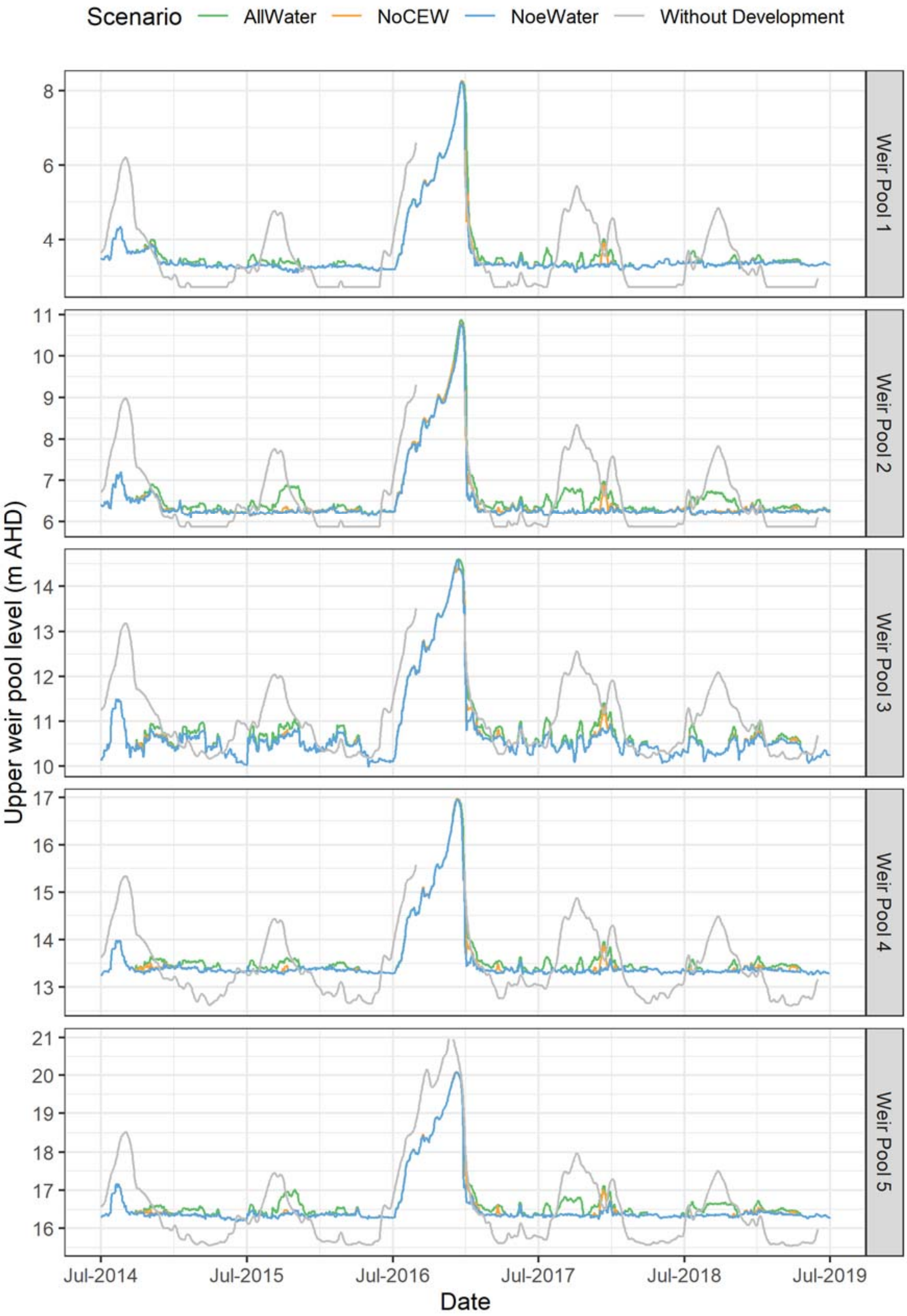

Figure 10. Modelled water level at the upstream end of each weir pool in the LMR.

Evaluation

To evaluate the contribution of Commonwealth environmental water towards an outcome, a contribution significance level was assigned to each evaluation question answer for a single year or five-year period. The level was viewed as 'to what extent Commonwealth environmental water contributed towards that observed outcome, with the ecological significance of the outcome considered where possible'. For example, the level assigned may be similar for an ecologically significant outcome of which Commonwealth environmental water had minor contribution towards versus an outcome that was considered negligible of which Commonwealth environmental water had major contribution towards. The thresholds for assigning the significance vary among indicators and questions, ranging from using arbitrary percentages or values, to qualitative assessment based on expert opinion.

Table 4. Hydrological Regime evaluation questions and answers. CEW = Commonwealth environmental water, eWater = environmental water.

CEWO evaluation questions	**Outcomes of CEW delivery**				
	2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to hydraulic diversity within weir pools?	13 km, 4% (17 km, 5%)	18 km, 5% (22 km, 6%)	20 km, 6% (53 km, 15%)	36 km, 10% (49 km, 14%)	An additional 15 km or 4% of lotic conditions created by CEW for at least 30 days (An additional 19 km or 6% for at least 14 days)
	CEW provided minor contributions to increased flowing water habitat in the LMR over the last five years, except for 2017-18, when CEW had a moderate contribution towards increasing lotic habitat (an additional 36 km (10%) for 30 days). Length of river with lotic conditions (velocity >0.3 m/s) has been used to represent hydraulic diversity. If there is some fast flowing water, it is expected there will be greater hydraulic diversity due to changes in habitat complexity (bends, backwaters, benches, etc.). Length of river with lotic conditions exceeded for 30 days over the year is presented, and 14 days in brackets. The time periods represent a number of flow-related ecological or life-history processes that could occur over periods of ~2–4 weeks. Total length of river assessed in the LMR = 345 km.				

<table>
<tr><th rowspan="2">CEWO evaluation questions</th><th colspan="5">Outcomes of CEW delivery</th></tr>
<tr><th>2014-15</th><th>2015-16</th><th>2016-17</th><th>2017-18</th><th>2018-19</th></tr>
<tr><td rowspan="2">What did CEW contribute to hydraulic diversity within weir pools?</td><td>13 km, 4% (17 km, 5%)</td><td>18 km, 5% (22 km, 6%)</td><td>20 km, 6% (53 km, 15%)</td><td>36 km, 10% (49 km, 14%)</td><td>An additional 15 km or 4% of lotic conditions created by CEW for at least 30 days (An additional 19 km or 6% for at least 14 days)</td></tr>
<tr><td colspan="5">CEW provided minor contributions to increased flowing water habitat in the LMR over the last five years, except for 2017-18, when CEW had a moderate contribution towards increasing lotic habitat (an additional 36 km (10%) for 30 days).
Length of river with lotic conditions (velocity >0.3 m/s) has been used to represent hydraulic diversity. If there is some fast flowing water, it is expected there will be greater hydraulic diversity due to changes in habitat complexity (bends, backwaters, benches, etc.). Length of river with lotic conditions exceeded for 30 days over the year is presented, and 14 days in brackets. The time periods represent a number of flow-related ecological or life-history processes that could occur over periods of ~2–4 weeks. Total length of river assessed in the LMR = 345 km.</td></tr>
<tr><td rowspan="2">What did CEW contribute to variability in water levels within weir pools?</td><td>0.10 m</td><td>0.15 m</td><td>0.08 m</td><td>0.17 m</td><td>Interquartile range (IQR) in water level increased by 0.10 m with and without CEW in the tailwaters (i.e. just downstream of each weir) across Weir Pools 1–5.</td></tr>
<tr><td colspan="5">CEW increased water level variability (IQR) in the tailwaters of weir pools during each year of the assessment, with the greatest increase in variability in 2017-18.
IQR is a measure of variability, as the difference between the 75th and 25th percentile values for water level over the year. If the IQR increases, the variability must have increased.</td></tr>
</table>

CEWO evaluation questions	Outcomes of CEW delivery				
	2014-15	2015-16	2016-17	2017-18	2018-19
What did CEW contribute to hydrological connectivity?	28 ha 20%	963 ha 37%	0 ha 5%	831 ha 39%	Maximum area inundated 484 ha greater with eWater. 20% increase in total volume of water for the water year at the SA border due to all eWater additions.
	CEW contribution to connectivity has been assessed laterally, as the increase in maximum inundated area each year, and longitudinally, as the percentage increase in volume each year (as used for Basin Scale assessment). CEW contribution to lateral and longitudinal hydrological connectivity was variable over the 5 years. Increases in area were predominately produced by weir pool raisings, supported by CEW, with moderately greater areas in 2015-16 and 2017-18. However, the lower area in 2018-19 was due to lower discharge during that weir pool raising event. The Basin-wide eWater strategy includes a target for a 30% increase in flows in the River Murray, which was met in 2 of the 5 years due to CEW.				

Contribution (to what extent CEW contributed towards the outcome, with the significance of the outcome considered):
⬜ Unknown 🟥 Negative ⬜ None/negligible 🟩 Minor 🟩 *Moderate* 🟦 Substantial

Discussion

A range of metrics has been considered to assess the evaluations questions in Table 4. Relevant velocity thresholds, proportions of the river, time of year and duration required for different ecological processes to be promoted are the focus of further research, as the empirical evidence relating the conditions occurring, and different ecological processes being promoted, continues to grow. It is expected that this hydraulic information, and the methodology developed to derive it for the future, will help to develop eco-hydraulic relationships. Noting this limitation in the current understanding of hydraulic requirements to promote desirable ecological functions and processes, the results are further discussed below.

The evaluation of Commonwealth environmental water contribution to hydraulic diversity within weir pools in the previous section indicates that the increase in discharge to 17,840 ML/d in 2017-18, compared to ~12,000 ML/d in other years (excluding the 2016-17 high flow year), increased the total number of river km with lotic velocities created by Commonwealth environmental water by a factor of 2–2.8 (Table 4, an increase from 13-18 km to 36 km in 2017-18). The magnitude of increase in 2017-18 compared to other years is also clear from Figure 7, and to a lesser extent Figure 8. This result highlights the substantial increases in lotic habitat in the LMR by increasing flow to South Australia in the order of 20,000 ML/d.

In years with some unregulated flow to build on, increases in lateral connectivity with Commonwealth environmental water in the LMR under current constraints to flow delivery in the southern connected Basin is largely limited to using infrastructure operations. Weir pool raising produced increases in inundated area as seen in Table 4 and Figure 7, with similar operations in the three years the raising events supported by Commonwealth environmental water. The differences in inundated area are produced by the differences in discharge during the raising events, with discharge of approximately 10,000 ML/d in 2015-16 and 2017-18 during raising events, compared to approximately 6,000 ML/d in 2018-19. Increases in discharge will increase water levels in the tailwater of the weir pool, and contribute to an overall greater inundation extent for the same weir pool height.

Commonwealth environmental water, and other held environmental water, has increased longitudinal connectivity throughout the LMR over the five-year period. With the exception of the 2016-17 high flow year, the only unregulated flow events occurred at the start of 2014-15 and a short event in December 2017. Without environmental water, flow to South Australia would have been at South Australian Entitlement levels during the rest of the five-year period, the minimum flow to be delivered to South Australia under Clause 88 of the MDB Agreement. The provision of some environmental water over most months in the period considered (as seen in Figures 5 and 7), has increased longitudinal connectivity through the system, including maintaining flow and fish connectivity over the barrages to the Coorong over the whole five year period.

There was limited ability to deliver environmental water in the first half of 2016-17 (as well as other times, for example summer of 2018-19), with the increases in velocity and water level (and in turn connectivity and hydraulic diversity) largely provided by unregulated overbank flows. Environmental water was provided to mitigate poor water quality conditions (low dissolved oxygen) and slow the recession in water levels following the flow peak. Despite this, water levels receded very quickly following the event (Figure 10). However, the modelling suggests this was in line with rates of fall/recession for natural conditions. Storages in the southern connected System were largely at full supply level, hence the steep recession was primarily driven by the fast recession in inflows to the system.

Management implications

The hydraulic model outputs underpinning the results have been presented in Figures 11 and 12 to help inform changes that can be expected from environmental water delivery and weir pool management. From these plots, the change in inundated area and proportion of the reach with lotic conditions (velocity >0.3 m/s) for each weir pool can be determined.

The inundated area can be seen in Figure 11, with the more horizontal the contour line, the greater the influence of weir pool raising on inundation extent. As discharge increases, the contour lines can be seen to slope to be more vertical, indicating that increases in discharge have a larger effect on inundated area, as the height of tailwaters of weir pools increase, with weirs close to overtopped by the maximum discharge shown of 45,000 ML/d (depending on the weir pool).

It is not just the maximum inundated area, or area inundated for a given duration that can enhance ecological responses, but also the variability in water levels. Variable water levels, and the coinciding periods of exposure and submergence of substrates beyond the euphotic zone, can result in the regular "re-setting" of biofilms (Steinman and McIntire 1990). The biofilm is a key component of riverine food webs, and this re-setting of the biofilm algal community produces higher quality food resources dominated by diatoms and unicellular algae (Wallace and Cummings 2016). Wallace and Cummings (2016) assessed biofilm changes during and following a 0.54 m raising of Lock 2 in 2015, and found only small changes in the biofilm composition directly upstream of Lock 2 immediately following the event, and no changes 35 days following the event. Based on this result, the authors suggested that frequent changes in weir pool level that mimic natural variability rather than annual "events" may be required to maintain early successional biofilm communities, expected to be a more nutritious food source.

The trade-off to increased inundation for the same discharge generated by raising water levels by operation of the weirs and other environmental water regulators is a reduction in velocity. This can be evaluated using Figure 12, showing the proportion of the weir pool with a velocity greater than 0.3 m/s for different combinations of discharge and weir pool manipulation. In Figure 12, the contour lines are close to vertical, indicating the weir pool manipulation has a marginal influence on the proportion of the weir pool exceeding this velocity threshold. Increases in discharge have a much larger influence on the proportion of the reach with velocity exceeding 0.3 m/s, with between 10–30% of the reach at a flow to South Australia of 10,000 ML/d, up to greater than 90% of the reach for flow of 30,000 ML/d. As noted earlier, further research is required to determine ecologically relevant metrics to interpret the consequences of changes in this particular reach scale metric.

The restoration of lotic habitats is important for ecological and life history processes for many native biota that are adapted to flowing riverine environments. For example, they provide stimuli for spawning of flow-cued species (e.g. golden perch) (King *et al.* 2016), facilitate downstream drift and transportation of plankton, macroinvertebrates and fish larvae (Gibbs *et al.* 2020), and provide diverse hydraulic habitats that are suitable for a range of species (e.g. Murray cod) (Zampatti *et al.* 2014). The reduction in the abundance and distribution of riverine (lotic) biota (e.g. Macquarie perch *Macquaria australasica* and Murray crayfish *Euastacus armatus*) throughout the MDB (Lintermans 2007) highlights the importance of restoring hydraulic conditions (e.g. lotic habitats), which is particularly needed in the heavily regulated LMR.

Conclusion

The five-year period evaluated was dominated by low flow conditions. With the exception of the 2016-17 high flow year, without environmental water the only flow events greater than the minimum flow to be delivered to South Australia under Clause 88 of the MDB Agreement occurred at the start of 2014-15 and a short event in December 2017. Under these minimum flow conditions, water levels would have been very stable throughout the year, with low hydraulic diversity and minimal fast flowing conditions in the LMR.

The provision of some environmental water over most months in the five-year period increased longitudinal connectivity through the system. Environmental water introduced some hydraulic variability that would not have otherwise occurred, to a greater degree in 2017-18 when discharge was increased to 17,840 ML/d compared to ~12,000 ML/d in other years (excluding the unregulated flow in 2014-15 and 2016-17 high flow year). In combination with weir pool raisings, environmental water also increased water level variability each year, as demonstrated by time series of modelled water levels with and without the environmental water, and the resulting interquartile range.

The understanding of the most relevant velocity thresholds, proportions of the river, time of year and duration required to promote different ecological processes is in its infancy. As the body of empirical evidence of these ecological responses increases, the hydraulic information presented in this section is expected to be foundational in understanding causal relationships, and in turn, the contribution environmental water has made to ecological outcomes.

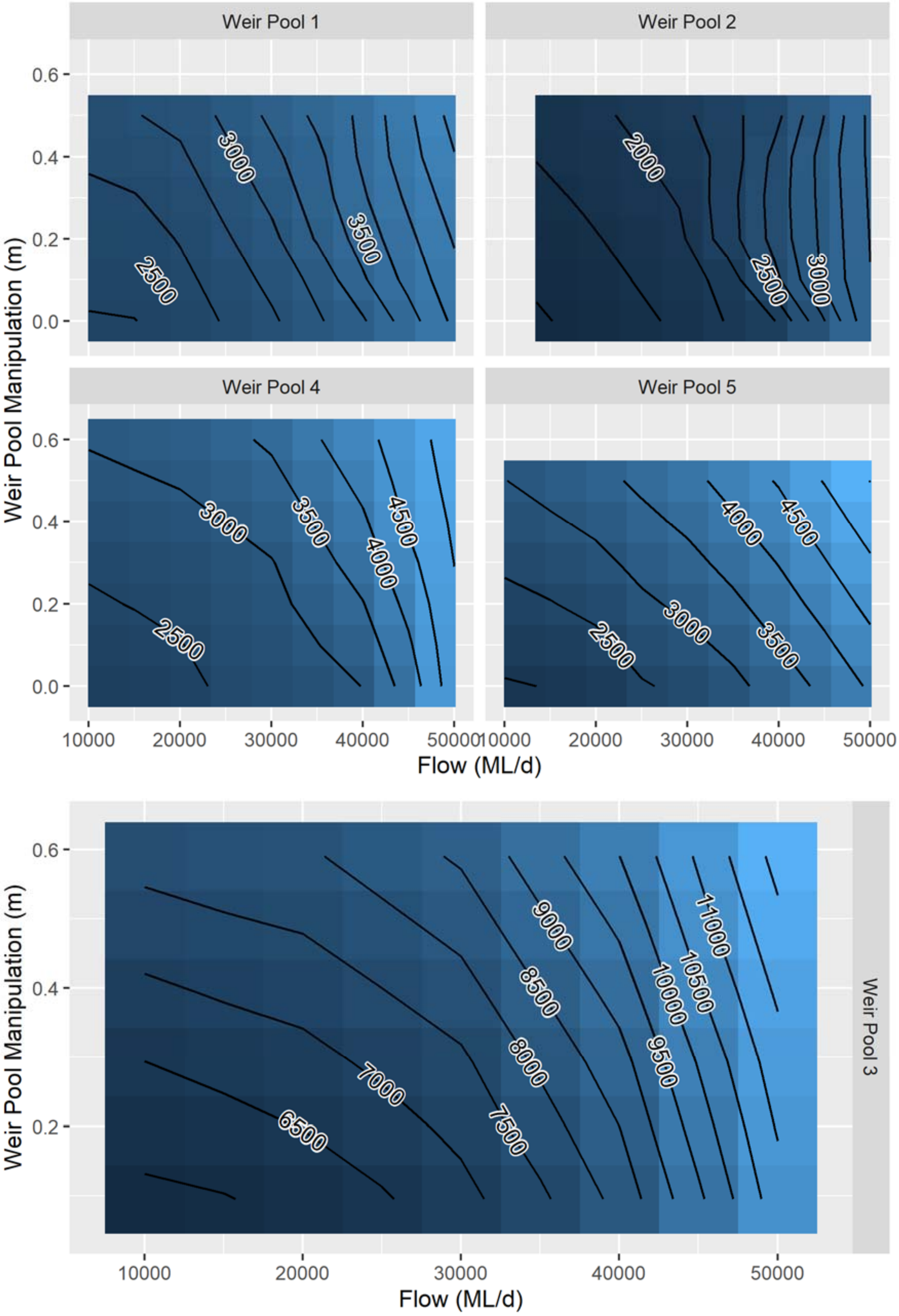

Figure 11. Area inundated (hectares) in each weir pool for changes in discharge and weir pool level in the LMR. Weir Pool 3 is shown on a different colour scale due to the larger areas in this weir pool.

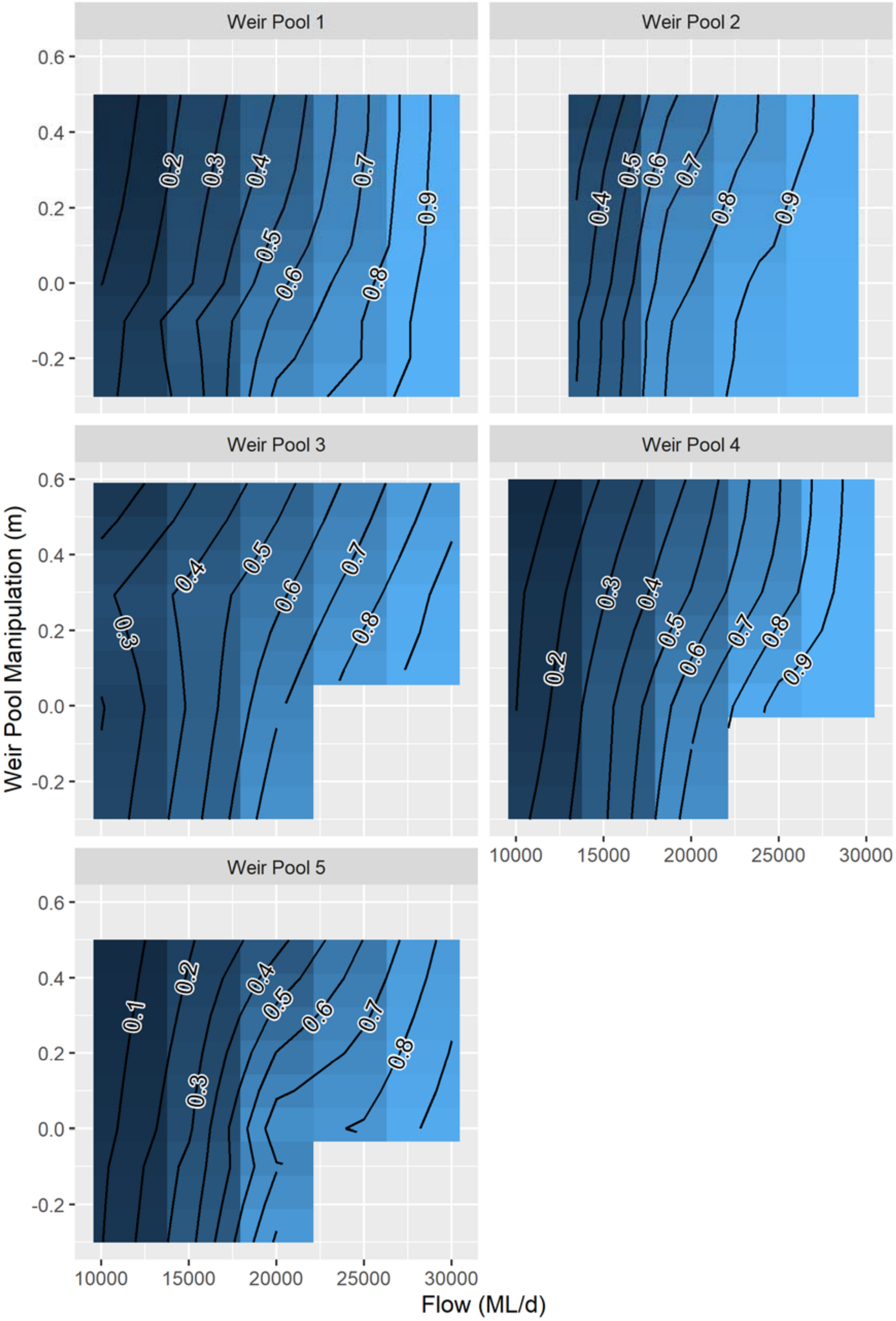

Figure 12. Proportion of the weir pool with velocities exceeding 0.3 m/s for changes in discharge and weir pool level in the LMR.

2.2 Stream Metabolism

Background

River food webs require organic materials as food resources. In deeper rivers with low light penetration, macrophyte growth is restricted and the microbiota (e.g. bacteria, protists and microalgae) form the lowest trophic levels of the food web. The incorporation of organic material by these organisms provides a major source of food for the higher trophic levels of microinvertebrates, macroinvertebrates and fish (Oliver and Merrick 2006).

There are two sources of organic carbon supplying the basal trophic levels. Photosynthetic microbes, comprised of microalgae and cyanobacteria, fix dissolved carbon dioxide using the energy of sunlight to form organic materials for cell growth. In deeper rivers this process is dominated by phytoplankton, the photosynthetic microbes that are suspended in the water column (Sellers and Bukaveckas 2003; Oliver and Merrick 2006; Várbíró *et al.* 2018). The availability of light for phytoplankton photosynthesis is influenced by the depth of light penetration relative to the average depth, and the intensity of water mixing that circulates phytoplankton through the upper illuminated surface layers. The availability and concentrations of nutrients also influence photosynthesis and the formation of organic materials (Reynolds 1984). Photosynthetic processes are affected by flow rates and water quality, and these are influenced by the catchment water sources, including environmental water contributions.

The second source of organic carbon is from external, terrestrial reserves (Berggren and del Giorgio 2015; Whitworth and Baldwin 2016). Terrestrial material can enter the river from wind movement, or from overhanging or fringing vegetation, but in floodplain rivers, like the Murray River, the transfer of terrestrial organic material is largely through flooding. Generally, this is comprised of materials that have accumulated on the floodplain between floods (Whitworth and Baldwin 2016). However, terrestrial materials that have accumulated on dry riverbanks and benches, or in fringing wetlands, can enter the river following increased channel flows that inundate these previously disconnected areas (McGinness and Arthur 2011). Flow patterns are important in determining the sources and supply of terrestrial organic carbon, and so environmental flows and their management will impact the carbon supply to the river food web (Oliver and Merrick 2006; Baldwin *et al.* 2016).

The respiratory breakdown of assimilated organic carbon provides energy and substrates for cell growth, with some associated loss of carbon as carbon dioxide. Although respiration is carried out by all river biota, the major contributors are the microbiota as they have higher respiration rates per unit mass and are usually present in greater quantities than organisms of larger size. Laboratory studies have shown that organic carbon recently fixed through photosynthesis is a primary source for phytoplankton respiration (Beardall and Raven 1990). In contrast, heterotrophic respiration is a result of the decomposition of organic materials sourced from the environment. Terrestrial organic carbon enters rivers in particulate and dissolved forms, but dissolved organic carbon is most actively incorporated by heterotrophic microbes such as bacteria and is a major driver of heterotrophic respiration (Graeber *et al.* 2018). Identifying the respiration associated specifically with phototrophs and heterotrophs is critical because they are both major contributors to the basal food resources (Graeber *et al.* 2018).

River metabolism measurements estimate the in-stream rates of gross primary production (GPP) and ecosystem respiration (ER), providing information on the sources and utilisation of organic carbon by riverine food webs (Odum 1956; Young and Huryn 1996; Oliver and Merrick 2006). Comparing rates of photosynthesis and respiration helps describe the fundamental trophic energy connections that characterise different food web types. It can indicate whether production or decomposition processes predominate, and whether the organic food materials have come from within the river (autochthonous sources) or from the surrounding landscape (allochthonous sources). The magnitude and characteristics of the metabolic processes indicate the size of the food web and its capacity to support higher trophic levels, including fish, which are key targets for ecosystem management (Odum 1956; Oliver and Merrick 2006; Oliver and Lorenz 2010; Sellars and Bukaveckas 2003).

Net ecosystem production (NEP), the difference between GPP and ER, is considered a measure of the overall carbon balance, and frequently used as an estimate of the basal food resource supply (Odum 1956; Young and Huryn 1996; Oliver and Merrick 2006). If GPP>ER, carbon is accumulating, while if GPP<ER, carbon is being lost from the system. However, this interpretation implicitly assumes that fixation of carbon through photosynthesis is the source of organic material fuelling respiration. This is not usually the case, as respiration is also due to the decomposition of allochthonous organic carbon by heterotrophs which results in heterotrophic production, an additional food resource not accounted for in the NEP calculation. Preferably the contributions of photosynthetic and heterotrophic organisms to net production can be estimated explicitly, as these are the two major pathways of organic carbon supply to the river food web.

During the five-year monitoring period, environmental water contributions to the flow in South Australia (Figure 5; Table 1), provided the opportunity to investigate their influence on metabolism. The objective of this study was to provide a more comprehensive assessment of metabolism in the LMR as a basis for evaluating the effects of environmental flows.

Major hypotheses

Increased flow (including the delivery of environmental water) into the LMR (peak and duration) in spring/summer will:

- Alter phytoplankton photosynthesis and the supply of autochthonous organic carbon to food webs if changes in channel flow volumes and water quality modify light and nutrient availability.
- Enhance ecosystem respiration (ER) rates and heterotrophic production if flows better connect the channel with riparian, wetland or floodplain areas, increasing the supply of allochthonous organic carbon.
- Reduce the likelihood of low dissolved oxygen (DO) concentrations by increasing water mixing in otherwise low flow zones except if flows carry excessive loads of organic carbon.

Methods

Field sampling

Rates of stream metabolism were estimated from changes in concentrations of dissolved oxygen, which also provided information on the suitability of DO to support the aquatic biota. Monitoring consisted of *in situ*, continuous ten-minute interval logging of DO, water temperature, and incident light at two river sites. One site was downstream of Lock 1 ('Lock 1' herein) in the gorge geomorphic zone, and one downstream of Lock 6 ('Lock 6' herein) in the floodplain geomorphic zone, refer to SARDI *et al.* (2018). Monitoring generally occurred from September to February/March each year (2014–2019), with occasional interruptions (< one day) during probe maintenance, except for the period 27 December 2015 to 20 January 2016 when data were not available for Lock 6 due to probe failure.

Two metre, depth-integrated water samples were collected during probe maintenance field trips (fortnightly to monthly), and analysed for chlorophyll *a*, total nitrogen, combined nitrate and nitrite, ammonium, total phosphorus, dissolved forms of phosphorus, and dissolved organic carbon. The detailed monitoring and analytical protocols described in Hale *et al.* (2014) were followed, but with some minor adjustments as detailed in Ye *et al.* (2018). In addition, the vertical light attenuation for Photosynthetically Active Radiation (PAR) was measured on each occasion using LiCOR underwater sensors.

Water quality measurements and vertical attenuation coefficients were considered relevant for two days before and two days after sampling, and this extended data set (220 points) was used to explore relationships with metabolic rates. In some cases, water quality data were supplemented with monitoring data from nearby sites provided by the Australian Water Quality Centre of South Australia Water.

Estimating metabolic rates

Daily volumetric rates for GPP and ER were estimated over 24-hour periods from midnight to midnight with the *BASE* program (Grace *et al.* 2015). This uses Bayesian regression routines to fit the measured changes in dissolved oxygen concentrations to a widely applied model that describes the daily fluctuations in water column dissolved oxygen concentrations (Odum 1956; Young and Huryn 1996; Oliver and Merrick 2006). The Levenberg-Marquardt curve fitting algorithm was also used to estimate metabolic rates as under some conditions it was computationally more efficient (Oliver and Merrick 2006; Oliver and Lorenz 2010). In general, both methods provide equivalent estimates of metabolic rates.

Oxygen based metabolic rates were converted to carbon units by assuming that the photosynthetic and respiratory quotients were equal to one. That is, every mole of oxygen transfer was considered matched by an opposite transfer of a mole of carbon dioxide.

The measured volumetric rates of metabolism were integrated over river depth and width based on channel morphometry. Daily flows measured during the monitoring periods were adjusted using hydrological modelling to estimate flows and water levels without Commonwealth environmental water, and without any environmental flows, and linked to channel morphometry. The morphometric data included the average cross-sectional areas, average depths, and average widths at different flow levels for river reaches stretching ca. 7 to 9 km upstream of each monitoring site. Currently, reliable estimates of water levels for flows modelled without environmental water are not available at the Lock 1 site. Water levels at this site are due not only to flow, but are complicated by interactions with Lower Lake levels and wind influences. Consequently, estimates of flow induced changes in metabolism are not presented for this site, but are expected to become available as modelling of water level progresses.

The volumetric metabolic rates were used to assess the influence of different water levels, while the cross-sectional metabolic rate, calculated as the product of the volumetric rate and the average cross-sectional area, was used to assess the effect of different flow volumes. The influence of flow on the average depth and cross-sectional area is a function of the channel morphometry, and increased flows may increase or decrease these depending on channel shape. Volumetric rates are reported as gm/m^3/day (numerically equal to mg/L/day), area as m^2 and so cross-sectional rates are for a 1 m length of river. Metabolic rates were integrated over time by summing daily rates to provide a basis for annual comparisons.

The influence of light on photosynthesis was analysed using the mean light intensity encountered by phytoplankton as they are mixed through the water column. The mean intensity depends on the incident irradiance (Io), the vertical attenuation of light passing through the water column (kd), and the average depth (zave). If the average depth is greater than the depth of the illuminated surface layer, then the mean irradiance (Im) is given by (Oliver and Merrick 2006):

$$Im = \frac{Io}{kd * zave} \qquad (1)$$

The amount of phytoplankton present was estimated from the chlorophyll concentration. Chlorophyll was converted to carbon by assuming it comprised 2.5% of cell ash-free dry weight and that cell carbon comprised 50% of ash-free dry weight, resulting in a multiplier of 20. Rates of GPP were standardised to chlorophyll concentrations in carbon units to account for differences in phytoplankton concentration (GPP(b)) and compared with the corresponding mean irradiances Im. The relative changes in GPP(b) due to Commonwealth environmental water, and total environmental water were calculated from the ratio of the average changes in depths resulting from these flows. Conversion of modelled GPP(b) estimates to GPP requires knowledge of the chlorophyll concentration. If it is assumed that the chlorophyll concentration on each day remains the same with and without environmental flows, then relative changes in GPP(b) and GPP are equivalent.

The contributions to ER by photosynthetic and heterotrophic microbiota was investigated using multiple regression. The potential component contributions to ER were estimated from a series of measures of respiratory sources. Chlorophyll-a provided a measure of the phytoplankton concentration, which has previously been shown to estimate their "maintenance" respiration when concentrations are high (Oliver and Merrick 2006). GPP provided a measure of the organic carbon produced by photosynthesis, which is a major source of phytoplankton respiration (Beardall and Raven 1990). DOC concentrations provided a measure of the resource supply for respiration of heterotrophic microbes. Components with insignificant regression coefficients were sequentially removed.

Respiration rates were averaged for each two-day period before and after the sampling date and compared with similarly averaged GPP rates, as respiration is influenced by GPP of the preceding day. Respiration rates were standardised to 25^0C before analyses to account for the influence of temperature.

The overall NEP was calculated as the difference between volumetric GPP and ER. The phytoplankton net production (PNP) was calculated as the difference between the measured GPP and the estimated phytoplankton community respiration (PCR). Similar calculations could not be used to estimate bacterial net production (BNP) as metabolism measurements do not provide a measure of bacterial gross production. Instead estimates were based on the bacterial growth efficiency (BGE), which is the ratio of the BNP to the sum of BNP and the bacterial respiration (BCR). Re-arrangement of this relationship provides an approach for estimating BNP but requires knowledge of the BGE:

$$BNP = \frac{BGE * BCR}{1 - BGE} \qquad (2)$$

The BGE is influenced by a range of environmental conditions including the chemical composition of DOC. However, an average value of 0.2 was considered reasonable for the Murray River within its typical temperature range and water quality attributes (Marra and Barber 2004; Rivkin and Legendre 2001; Berggren and del Giorgio 2015). The combined net production (CNP) was estimated as the sum of the calculated PNP and BNP.

Results

Daily metabolic rates

Patterns of daily GPP (photosynthesis) and ER (respiration) varied markedly within and between years, and across sites, with particularly high respiration rates evident in the 2016-17 flood year (Figure 13). Generally, GPP and ER were of similar magnitude and daily NEP values varied between negative and positive values but were often close to zero, with integrals over time close to zero. Understanding the causes of these patterns and interpreting their significance to food resource supply is a key objective of the following analyses.

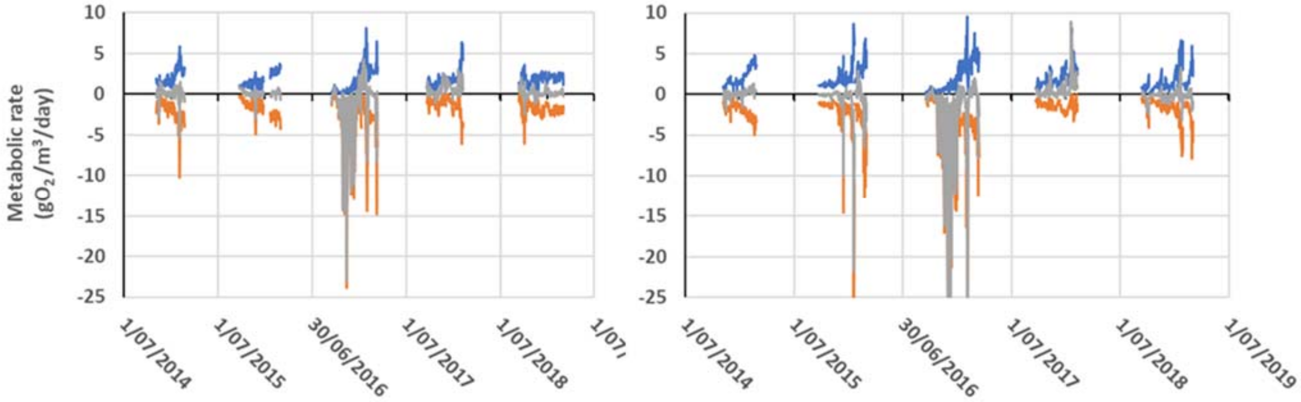

Figure 13. Rates of GPP (●), ER (●) and NEP (●) at (left) Lock 6 and (right) Lock 1 over the five monitoring periods from 2014–2019.

Gross primary production

There was a strong correlation between GPP(b) and *Im*, although the variation was large, especially at the Lock 1 site. Previous metabolism measurements in the weir pools showed that at water velocities above ca. 0.2 m/s mixing entrained the phytoplankton and *Im* reliably described the light encountered (Oliver and Lorenz 2010). Re-analysing data for water velocities ≥0.22 m/s greatly improved the regression relationship, with GPP(b) proportional to *Im* (Figure 14).

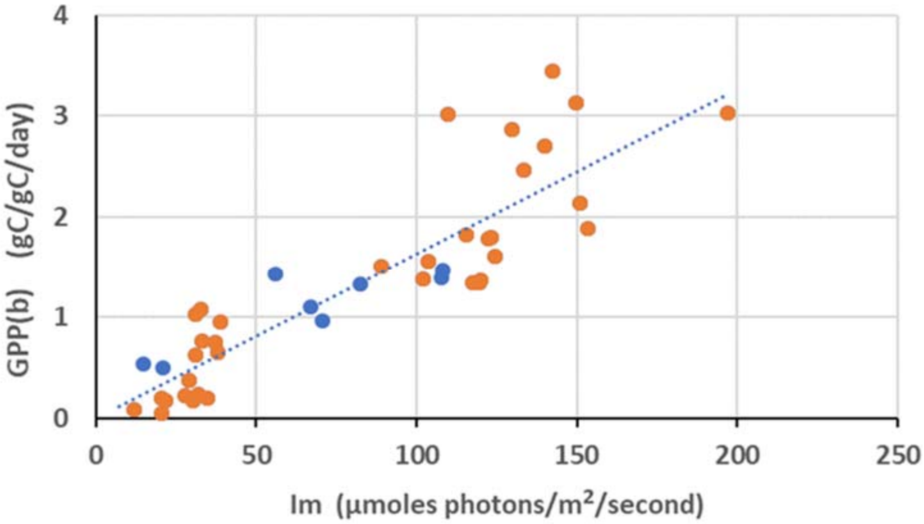

Figure 14. The response of GPP per unit phytoplankton biomass in carbon units, to the mean irradiance of the water column over the daylight period for Lock 6 (orange circles) and Lock 1 (blue circles) when water velocities were greater than 0.22 m/s. Regression, y = 0.0164x - 0.0024, r^2= 0.78.

Combining this regression with Equation 1, GPP can be estimated from:

$$GPP = GPP(b) * chla = 0.016 * \frac{Io}{kd * zave} * chla \qquad (3)$$

This indicates that increases in the average depth reduce the volumetric rate of GPP(b) in inverse proportion to the relative change in depth. At Lock 6, the increases in flow due to environmental water (Figure 15A) generally reduced the volumetric GPP(b) as average depths increased (Figure 15B). The predicted change in GPP(b) ranged from an increase of 3% to a reduction of 28%. If water quality, including the chlorophyll concentration and kd, remain unchanged for the daily comparisons of flows with and without environmental water, then fractional changes in volumetric GPP are equivalent (Equation 3).

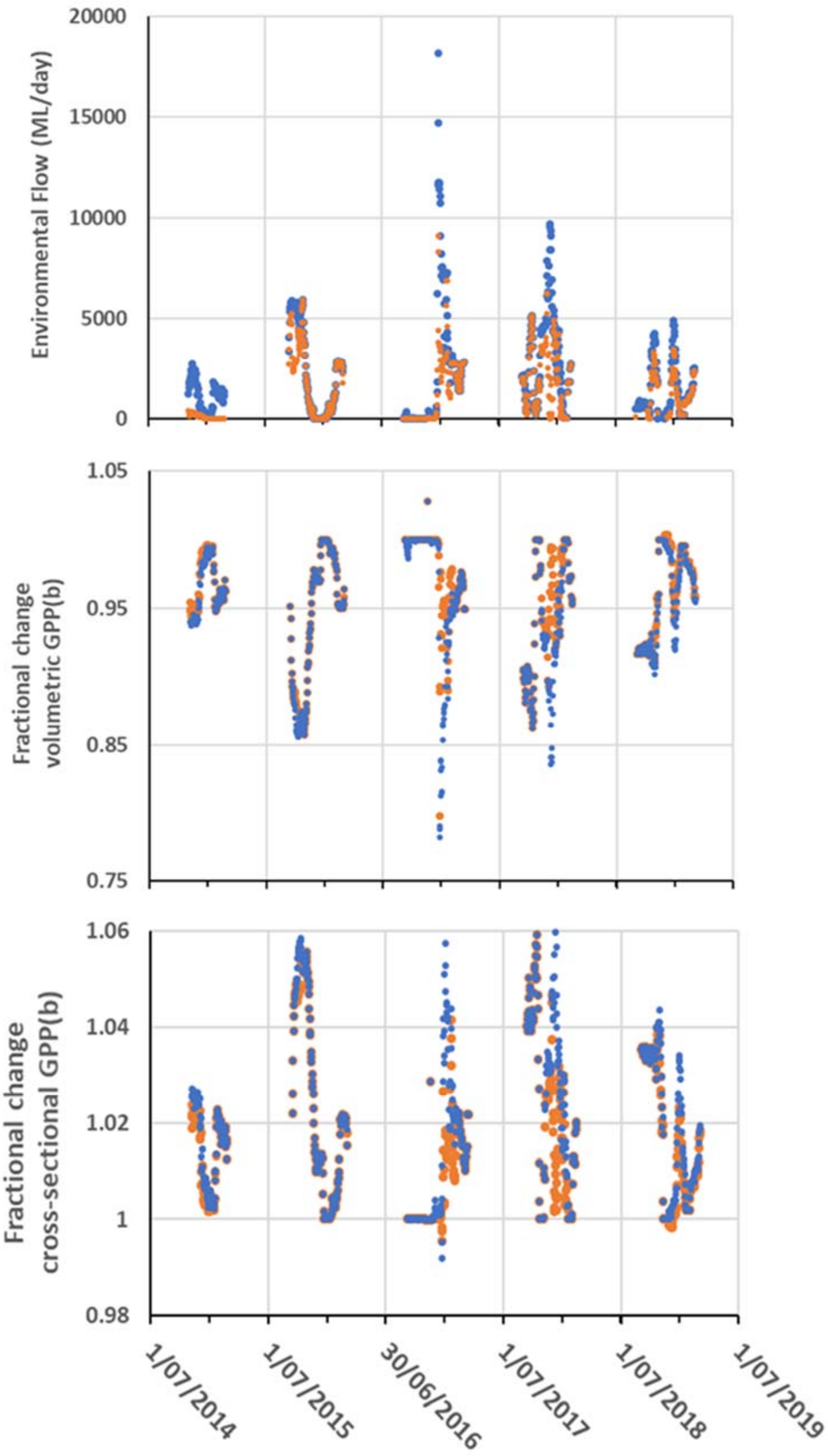

Figure 15. (A) Environmental water contributions to discharge at Lock 6, (B) Fractional change in volumetric GPP(b), (C) Fractional change in cross-sectional GPP(b), due to flow including all environmental water (●) or only CEW (●).

In contrast, environmental flows were generally associated with increases in the cross-sectional area at Lock 6, leading to increased rates of daily river production per metre of river (Figure 15C). The change in cross-sectional GPP(b) with environmental water ranged from a reduction of 1% to an increase of 6.5%. Similar relative changes in volumetric and cross-sectional GPP(b) were estimated for Lock 1.

Integrated Gross Primary Production

At Lock 6, the addition of environmental water reduced the volumetric GPP integrated over the monitoring period by between 5 and 21.5 gO_2/m^3/monitoring period (Figure 16). These are small changes relative to the accumulated GPP without environmental flows, representing relative changes of 0.94 to 0.98 of accumulated rates without environmental flows, reflecting the small reductions in daily volumetric rates (Figure 15).

Increases in cross-sectional GPP due to the addition of environmental water were also small (Figure 15) and the accumulative outcome gave increases of between 637 and 2,445 gO_2/river m/monitoring period (Figure 16). These represent increases of between 1.01 and 1.02 of the accumulated rates without environmental flows.

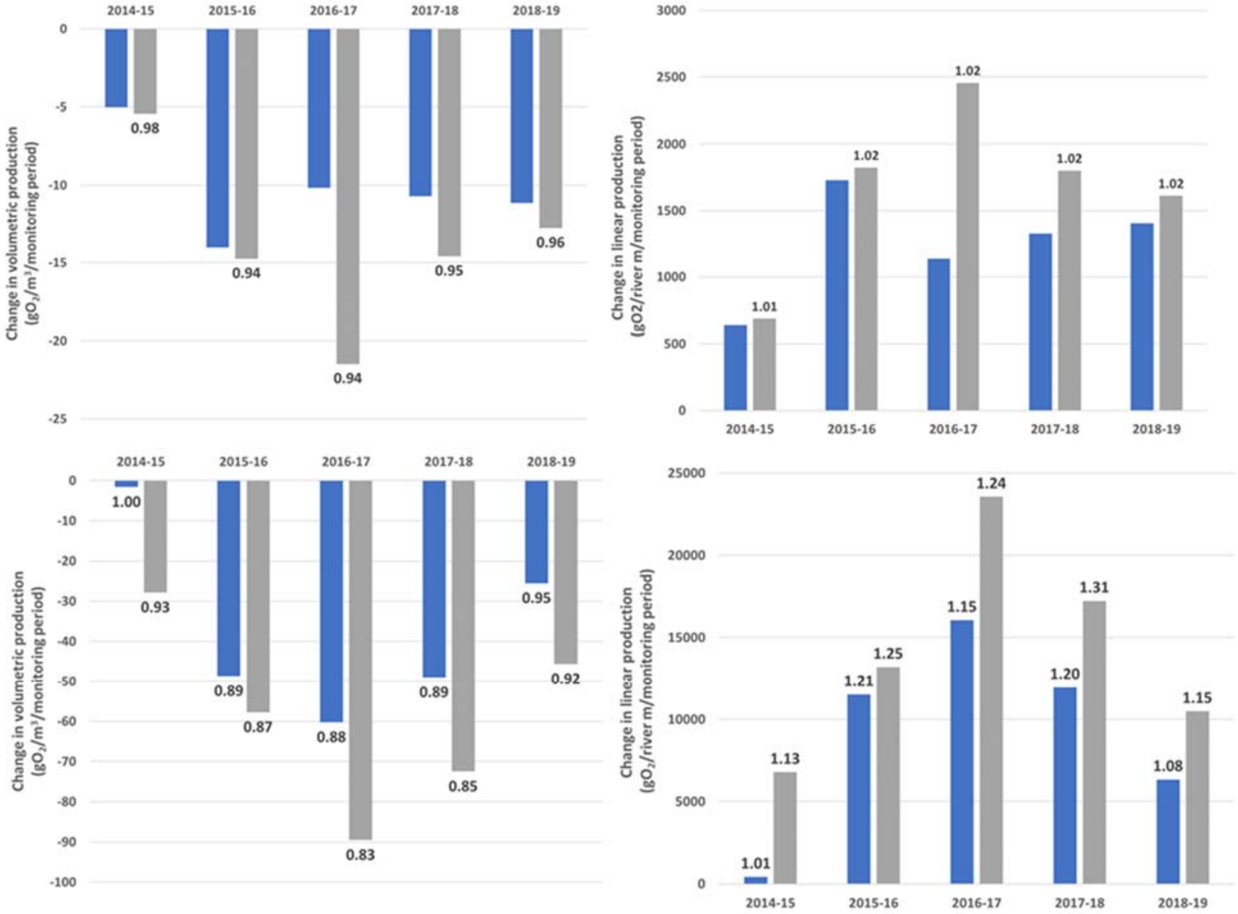

Figure 16. Flow induced changes at Lock 6 (top) and Hattah (bottom) in (left) cumulative volumetric GPP over each monitoring period, (right) cumulative cross-sectional production over each monitoring period, due to the addition of CEW (blue bars) and all environmental water (grey) from no environmental water. Fractions above each bar represent proportional changes in GPP due to environmental flows compared with no environmental flows (e.g. 1.02 indicates GPP increased by 2% over no environmental water).

These small changes in GPP in response to flow reflected the stable water levels within the weir pool systems. To demonstrate the scale of effects that might occur in unregulated river reaches, the responses were modelled for Hattah, a site previously monitored and for which channel morphometry was available (Oliver and Merrick 2006). Modelling assumed the same flow regimes as observed at Lock 6, and utilised the relationships between flow, average depth and cross-sectional area at the Hattah site. Due to the greater shifts in water level at this site, the modelled changes in GPP were significantly larger than at the Lock 1 and 6 monitoring sites (Figure 16). The volumetric GPP reduced by between 0 and 90 gO_2/m^3/monitoring period, representing reductions in rates to 0.83 of those without environmental flows. Similarly, the changes in cross-sectional GPP were larger with increases of between 0 and 23,600 gO_2/river m/monitoring period, representing rates of up to 1.31 times those without environmental flows (Figure 16).

Vertical attenuation of light

The light available for photosynthesis is influenced not only by the average depth but also the depth of light penetration. This is determined by kd (Equation 4), which depends on the rate at which light is absorbed by coloured material and scattered by particles as it passes through the water (Kirk and Oliver 1995; Oliver *et al.* 2010; Cottingham *et al.* 2010). A multiple regression analysis indicated that DOC and turbidity (NTU) accounted for 92% of the variation in kd, described by the relationship:

$$kd = 0.224 * DOC + 0.0577 * NTU + 0.1246 \qquad (4)$$

with standard errors for coefficients of 0.013, 0.003, and 0.16 respectively. Values for kd calculated from the regression were closely aligned with measured values (Figure 17). Consequently, any change in flow, or water source that alters the DOC concentration or turbidity, will alter the mean irradiance encountered by phytoplankton and affect GPP(b).

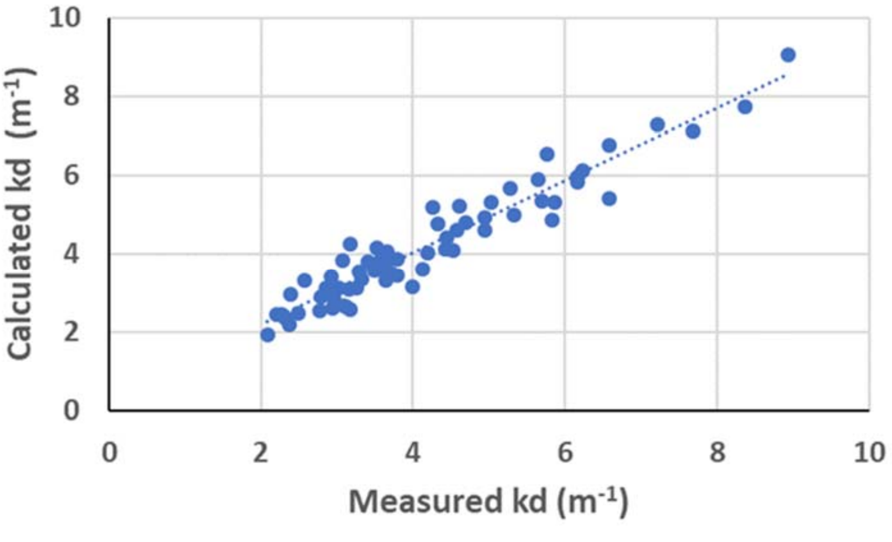

Figure 17. Comparison of the vertical attenuation coefficient kd measured in the field and calculated from the dissolved organic carbon concentration and the turbidity.

DOC and turbidity

The large differences in GPP between years were not explained solely by changes in depth and cross-sectional area, indicating that other environmental influences were at play. No strong relationships were observed with nutrient levels, but the three-fold changes in kd impacted significantly on GPP(b) which is inversely related to kd (Equation 3). Turbidity and DOC both affect kd (Equation 4) and comparing their contributions demonstrated the importance of turbidity in influencing GPP(b) (Figure 18). Similar results were obtained at the Lock 1 site.

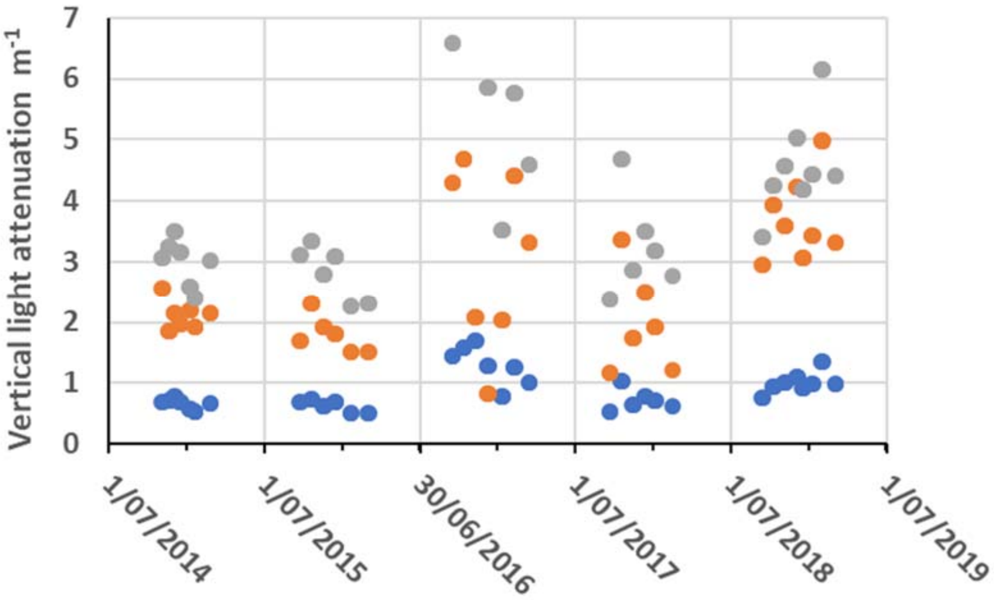

Figure 18. The vertical attenuation coefficient for Photosynthetically Active Radiation (●) and the contributions due to DOC (●) and NTU (●) at Lock 6 for each monitoring period.

Ecosystem and Community Respiration

A significant correlation was obtained between ER and the variables GPP and DOC (r^2 = 47%, n = 96, GPP Coefficient 0.74 (± 0.12); DOC Coefficient 0.09 (± 0.01)). The regression included a small intercept that was not-significantly different from zero (*P*<0.01). These coefficients were used to provide a comparison of respiration rates due to phytoplankton and bacteria (Figure 19). In most monitoring periods the bacterial contribution to ER (BCR) was equivalent to, or less than the phytoplankton contribution (PCR), except early in the 2016-17 flood year when BCR made up almost all the respiratory activity.

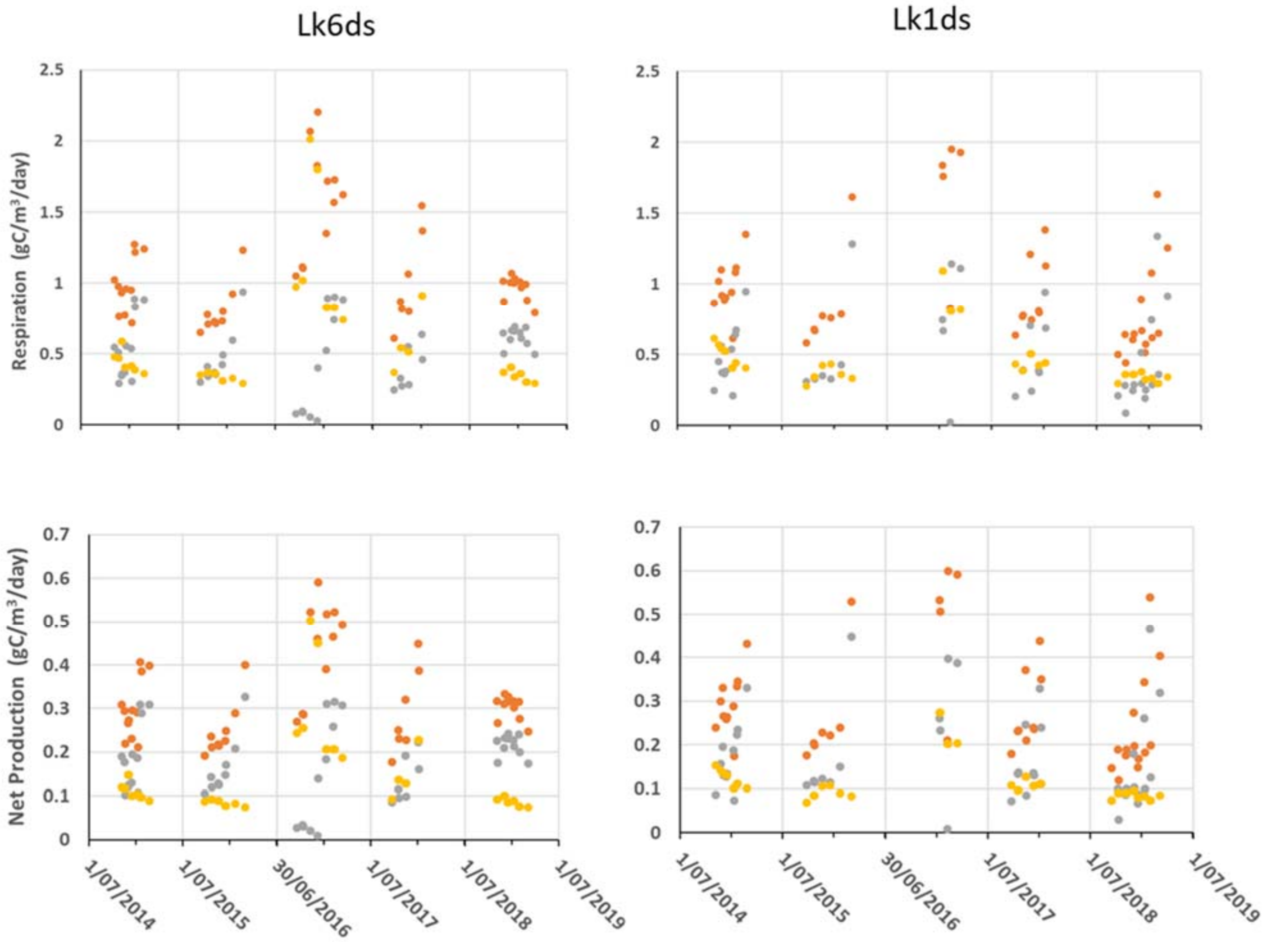

Figure 19. Respiration rates and net production for phytoplankton (●), bacteria (●) and both combined (●).

Using these respiration rates, net production was estimated for the phytoplankton and bacteria separately and showed large differences between years (Figure 19). At both sites values for CNP were positive and ranged between 0.1 and 0.6 gC/m^3/day, with data clustered around a mean across both sites of 0.3 (± 0.03, 95%CL) gC/m^3/day (Figure 19). The phytoplankton contribution to the CNP at Lock 6 varied between 40 and 80%, except early in the 2016-17 flood year when the contribution was reduced, ranging from 0 to 50%. At this time much larger contributions were attributed to bacterial net production associated with high respiration of DOC contained in flood waters. Similar results were obtained at Lock 1 (Figure 19).

Ecosystem Respiration, Combined Net Production and altered flows

Estimates of ER for the modelled flow conditions without Commonwealth environmental water or environmental water need to include the contributions from both phytoplankton and heterotrophs. Phytoplankton respiration was correlated with GPP, so the same assumptions were used as in estimating GPP under the different flow scenarios. These were, that on any day the chlorophyll concentration was the same with and without environmental water, and that water quality was largely unchanged, especially turbidity and DOC that influence light attenuation. Under these assumptions the relative changes in the phytoplankton respiration (PCR) will match the relative changes in GPP (Figure 16) for both volumetric and cross-sectional estimates.

The bacterial respiration (BCR) is correlated with the DOC concentration which was assumed to remain unchanged on any given day, enabling estimation of rates without environmental flows. In contrast to GPP, where flow induced changes in light availability influence the estimates of volumetric rate, if DOC concentration is unchanged then the volumetric BCR remains unchanged. Consequently, the linear river rates will alter in proportion to the relative changes in cross-sectional area. The effect of this can be estimated from Figure 16 by adjusting the relative changes in cross-sectional GPP rates for the relative changes in the volumetric rates. In the Lock 6 data the relative increases in BCR will be between 2 and 6% higher than the relative changes shown for cross-sectional GPP in response to environmental water, ranging between factors of 1.03 and 1.09. The modelled results for the Hattah site indicate relative changes in BCR between 7 and 17% higher than the relative changes shown for cross-sectional GPP, with factors ranging between 1.22 and 1.54, indicating substantial increases in the cross-sectional decomposition rate with the addition of environmental water.

The data indicate that phytoplankton net production is a constant fraction of GPP and so will change in proportion to GPP. Consequently, the proportional changes in GPP depicted in Figure 16 match the proportional changes expected in PNP, with increased flows reducing volumetric rates but increasing cross-sectional rates. The heterotrophic net production as estimated from Equation 2 is proportional to BCR and consequently heterotrophic net production increases proportionally to increases in BCR in response to flows.

If cross-sectional rates of NP reflect a "carrying capacity" for the river in terms of carbon production available for higher trophic levels, then at the unregulated Hattah site the modelled results indicate that in 2017-18 integrated cross-sectional PNP was increased by a factor of 1.31, and BNP by a factor of 1.54 as a result of environmental water flows.

Dissolved oxygen

Modelling of phytoplankton metabolism indicated that PNP was on average positive, generating a net oxygen surplus, while bacterial respiration of DOC reduced oxygen concentrations. The DO concentration resulting from this interplay depends not only on the light intensity and the concentration of DOC, but also on the gas exchange at the water surface. Reduced exchange rates enhance the possibility of oxygen depletion when DOC concentrations are high. The gas exchange coefficient was found to be a function of the water velocity, which along with wind generates mixing in the river (Figure 20A). At velocities below ca. 0.18 m/s, gas exchange was reduced to values approaching zero. This critical velocity was used to assess the influence of environmental flows in reducing the potential for low oxygen periods by increasing velocities and maintaining gas exchange rates. In most years, environmental flows made significant contributions, particularly during the spring and summer periods when respiration rates were high. The number of days in each monitoring period meeting the velocity criteria (0.18 m/s) without environmental water contributions, with Commonwealth environmental water, or with all environmental water, are presented in Table 5 and Figure 20B.

Table 5. Number of days in each monitoring period meeting the velocity criteria (0.18 m/s) without environmental water (eWater) contributions, with Commonwealth environmental water (CEW), or with all environmental water.

Year	Total number of days		Additional days due to	
	No eWater	All water	Other eWater	CEW
2014-15	0	31	31	0
2015-16	0	69	16	53
2016-17	130	165	14	21
2017-18	12	80	18	50
2018-19	1	46	20	25

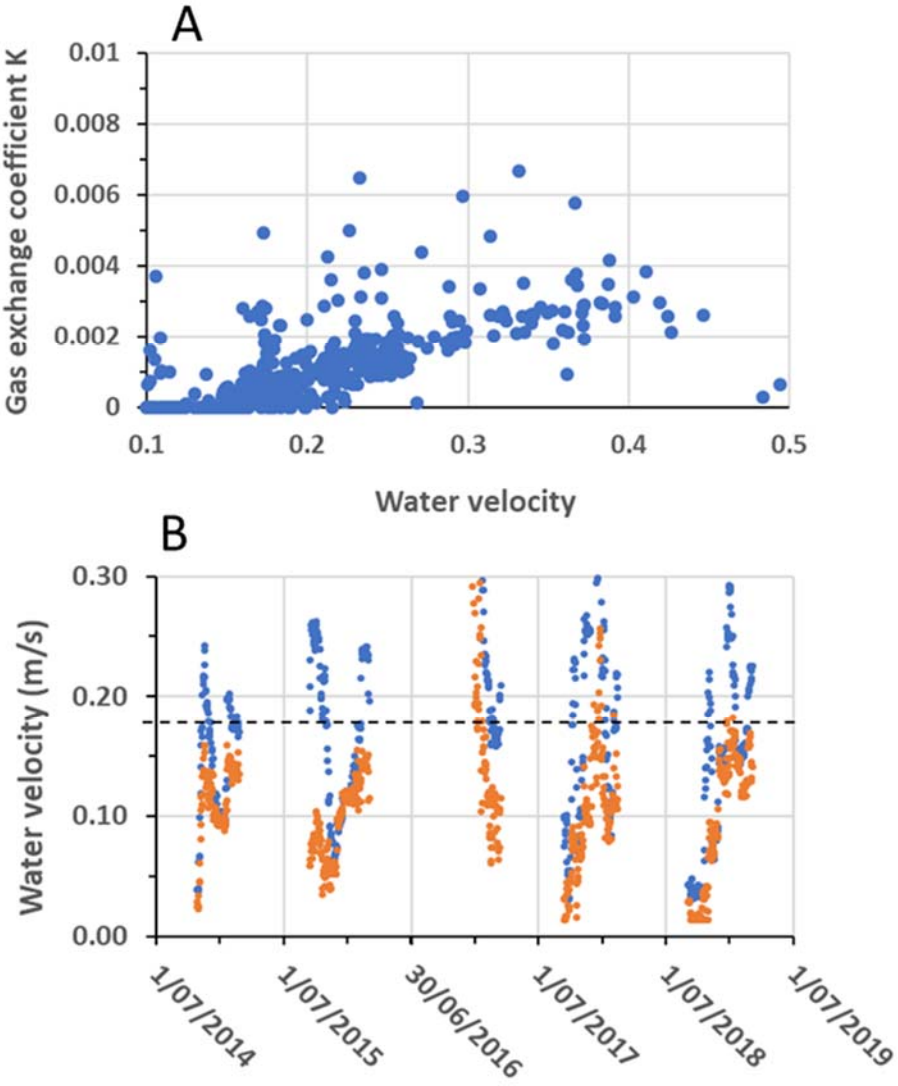

Figure 20. (A) The gas exchange coefficient at different water velocities and (B) the water velocities with (●) and without (●) environmental water at Lock 6.

Evaluation

The evaluation approach, including assessment criteria, is described in the evaluation section for hydrological regime (Section 2.1).

Table 6. Stream Metabolism evaluation questions and answers relating to Commonwealth environmental water (CEW) and environmental water (eWater). The Lock 6 site has been used to answer the evaluation questions.

CEWO evaluation questions	Outcomes of CEW delivery					
	eWater type	**2014-15**	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to dissolved oxygen levels?	All eWater	31	69	35	68	45 extra days with reduced risk of low DO
	CEW	0	53	21	50	25 extra days with reduced risk of low DO
	eWater decreased the likelihood of low DO by increasing water mixing and oxygen exchange at the surface. This was assessed as the extra days per year with water velocities > 0.18 m/s due to eWater. A substantial contribution was considered greater than 30 days, moderate 15-30 days, minor 7-14 days and negligible < 7days.					
What did CEW contribute to patterns and rates of primary productivity?	All eWater	1.01	1.02	1.02	1.02	1.02 fractional increase in cross-sectional GPP
	CEW	1.01	1.02	1.02	1.02	1.02 fractional increase in cross-sectional GPP
	Increased flows generally reduced the volumetric rate of primary production but increased the cross-sectional rate. This increased the overall "carrying capacity" of the river, although the implications of changes in the ratios of these two measures are unknown. At the LMR sites, the fractional increases in cross-sectional GPP due to eWater were negligible due to the largely fixed water levels set by weirs. A substantial contribution was considered an increase in cross-sectional GPP of 20% or greater, moderate 11-19%, minor 5-10%, negligible <5%.					

CEWO evaluation questions	Outcomes of CEW delivery					
	eWater type	2014-15	2015-16	2016-17	2017-18	2018-19
What did CEW contribute to patterns and rates of decomposition?	All eWater	1.03	1.09	1.09	1.07	1.06 fractional increase in cross-sectional BCR
	CEW	1.03	1.09	1.09	1.07	1.06 fractional increase in cross-sectional BCR
	Bacterial respiration (BCR), a measure of decomposition, is directly related to DOC concentrations. Modelling of the influence of flows on BCR assumed that for any given day DOC concentrations were the same with and without eWater. Relative changes in river linear BCR due to the addition of CEW at Lock 6 were relatively small due to the constant water level maintained by the weirs. A substantial contribution was considered an increase in cross-sectional BCR of 20% or greater, moderate 11-19%, minor 5-10%, negligible <5%.					

Contribution (to what extent CEW contributed towards the outcome, with the significance of the outcome considered):
⬜ Unknown 🟥 Negative 🟩 None/negligible 🟩 Minor 🟩 Moderate 🟦 Substantial

Discussion

The linear correlation between GPP(b), the gross primary production per unit of phytoplankton biomass, and the mean irradiance (Im) was comparable with those previously reported for the Murray River, supporting its reliability (Oliver and Merrick 2006; Oliver and Lorenz 2010). Using this relationship (Equation 3), in conjunction with measurements of channel morphometry, enabled estimation of volumetric and cross-sectional rates of GPP(b) with and without environmental flows. To convert biomass specific rates to GPP, the biomass of phytoplankton is required, but this is difficult to determine independently for the different flow conditions. Also, *Im* is influenced not only by changes in water depth but also by water quality, particularly turbidity and DOC concentrations, which could be influenced by the water quality of the environmental flows. It was difficult to identify and quantify the different sources of environmental flows delivered during the monitoring periods, and what the water quality conditions would have been like without the environmental flows. To assess the influence of flows it was assumed that daily modelled flows without environmental water had the same water quality and chlorophyll concentrations as the measured flows, so that relative changes in GPP(b) corresponded to the changes in GPP.

At the monitoring sites in the LMR, the effects of environmental flows on volumetric and cross-sectional GPP were small across all years due to the weirs that maintained relatively constant water levels (Figure 15; Figure 16). To indicate the potential effects of flow interactions with channel morphology, changes in GPP were modelled for a less regulated channel reach at Hattah, using the same flows observed at the Lock 6 site (Figure 16). At the Hattah site, GPP underwent large changes with up to 17% reductions in volumetric rates and 24% increases in cross-sectional rates. This demonstrated that the interaction of flows and channel morphometry can have a major influence on the volumetric and cross-sectional rates of GPP.

In addition to photosynthetic production, the food web is also supported by the supply of heterotrophic production through DOC utilisation. Rates of ER were partitioned into respiration by phytoplankton (PCR) and by the heterotrophic bacteria (BCR), and this also enabled individual estimates of their net production (Figure 19). In most monitoring periods the bacterial contribution to ER (BCR) through decomposition was of similar magnitude to the phytoplankton photosynthate respiratory contribution (PCR). Only during the flood in early 2016-17 was the BCR consistently larger than PCR as a result of high DOC concentrations in the flood waters (Figure 19). The combined net production (CNP) of these two components estimated their supply of organic carbon to the food web. This ranged between 0.1 and 0.6 gC/m^3/day, with a mean of 0.3 gC/m^3/day, providing a very different perspective on carbon supplies to the river food webs compared to the traditional analyses of NEP. In comparison, NEP fluctuated between negative and positive values that were often close to zero, with integrals over time also close to zero, as previously reported (Oliver and Merrick 2006; Gawne *et al.* 2007). The CNP estimates demonstrate that both heterotrophic and phytoplankton production are important sources of organic carbon to the river. Improved supplies of DOC will be critical to providing food webs with organic carbon food resources through the heterotrophic pathway, but an upper limit is set by the influence of DOC respiration on DO concentrations. Models assessing the level of DOC concentrations that lead to "black water" events are developing (Baldwin *et al.* 2016; Whitworth and Baldwin 2016), but a holistic approach that considers the full implications of river metabolism could further benefit management of environmental flows.

The effects of environmental flows on volumetric and cross-sectional rates of decomposition were estimated from the changes in bacterial community respiration (BCR). As with GPP, assumptions were necessary to estimate BCR in the absence of environmental flows. In this case, it was assumed that the DOC concentration on any day was the same with and without environmental water. Integrated rates of BCR over the monitoring periods provided a means for comparing annual differences with and without environmental flows. As BCR rates are a function of DOC, the assumption of equivalent DOC concentrations on any day meant that volumetric rates of BCR did not differ with flow. In contrast, relative changes in the cross-sectional BCR were equivalent to the relative changes in cross-sectional area with flow. Differences in river cross-sectional BCR were small at the monitoring sites in the LMR due to the weirs that maintained relatively constant water levels (Figure 15; Figure 16). In contrast, changes at the modelled Hattah site were large with relative increases of up to 1.54 due to increased flows.

Between year differences in metabolism were large and mainly related to changes in water quality, especially turbidity and DOC (Figure 13). These water quality differences could not be attributed specifically to environmental flows, and so further inter-annual analyses were not undertaken. However, they highlight the major impact that water quality has on river metabolism.

The dissolved oxygen concentration is influenced by water quality, particularly through the respiration of DOC by bacteria, but it is also influenced by flow, especially in low flow areas such as weir pools where surface oxygen exchange is reduced. Increased flows caused by environmental water can improve the DO conditions in low flow reaches by increasing mixing and enhancing oxygen exchange at the surface. At velocities above 0.2 m/s, phytoplankton GPP(b) is proportional to *Im* indicating that mixing is occurring (Figure 14). Also, the gas exchange coefficient is close to zero below 0.18 m/s and increases at higher velocities (Figure 20). These findings suggest that if water velocity can be sustained above 0.18 to 0.2 m/s then mixing will be enhanced and help minimise DO depletion, except when DOC concentrations are high. The contributions to flow from environmental water (including Commonwealth environmental water) made substantial contributions to improving gas exchange conditions in most years (Table 5) helping to avoid potential water quality impacts and associated potential for fish kills.

Analyses of the monitoring data have provided new and important insights into river metabolism and its significance in supplying food resources to the river ecosystem. In general, these findings support the hypotheses regarding the influences of flow on river metabolism and dissolved oxygen concentrations. Currently these findings are restricted to the monitoring sites, but the modelled relationships provide an opportunity to extend this understanding to the Murray River more generally, especially in conjunction with currently available hydrological models such as Source (Beh *et al.* 2019). Such collaboration holds significant promise to further our understanding of river metabolism and the influence of flows, including those due to environmental water.

Management implications

Increased flows interact with channel morphometry altering the average depth and the cross-sectional area of the flow and affecting the rate of volumetric and cross-sectional GPP. Depending on channel shape, changes in flow are likely to be more beneficial at some water levels in the channel than at others, particularly where the channel broadens. Using environmental flows to target these water levels could increase river productivity. Also, the manipulation of weir levels could alter metabolic conditions within weir pools, potential shifting the interaction between volumetric and cross-sectional GPP and altering carrying capacity. Further exploration of this potential is being explored using the developed hydrological and metabolic models.

In general, Commonwealth environmental water deliveries increased the average water depth and reduced volumetric GPP, but increased cross-sectional areas which increased the cross-sectional GPP. These opposite shifts in local food production versus total river food production are likely to have fundamental effects on the composition and functioning of food webs, but the significance is currently not understood.

GPP(b) relies directly on the mean light within the water column. In addition to average depth, the mean light also depends on light attenuation which is related to turbidity and DOC concentrations. Environmental flows which alter the attenuation of light through increased turbidity and DOC, can greatly influence GPP(b). However, DOC concentrations are important to heterotrophic metabolism, with increased concentrations enhancing heterotrophic net production. These opposite influences of DOC on phytoplankton and heterotrophic net production is one example of several trade-offs that need to be considered regarding the water quality of flows.

Bacterial respiration is a function of the DOC concentration and leads to bacterial production which enhances carbon supplies to the food web. However, high DOC concentrations can cause enhanced bacterial respiration rates leading to oxygen depletion. Environmental flows need to be managed to achieve beneficial DOC concentrations either by selecting appropriate sources of water supply from the catchment, or by managing flows to achieve suitable interactions with terrestrial supplies of organic carbon as flows progress downstream. The latter approach will need to consider the accumulation of terrestrial carbon on the floodplain, the area that might be inundated to provide a beneficial supply of DOC, without leading to concentrations that have detrimental effects, especially on DO concentrations.

Phytoplankton and bacteria both contribute to the basal net production supplying food resources to river food webs. The quantity, quality and characteristics of the food supply will be influenced by the contributions from each of these sources which flows and water quality influence through their effects on GPP and heterotrophic production. The food web requirements for the supply of these various resources are not well known but they are expected to influence the riverine community structure so a better understanding of the links with higher trophic levels is required.

Commonwealth environmental water can help reduce the likelihood of low DO concentrations in the LMR, if it increases water velocities above a critical level of ~0.2 m/s, below which surface oxygen exchange is poor. This critical velocity may vary with channel morphology and further investigation is required in other river sections where de-oxygenation has been a problem. The level of flow required to help offset de-oxygenation is also influenced by the DOC concentration, and if this is high then it may not be possible to offset with surface exchange.

Conclusion

The analyses identified key environmental influences on GPP and ER including: the reliance of GPP on the mean light encountered by phytoplankton; the effects on GPP of interactions between flow and channel morphometry; the reliance of the mean light on turbidity and DOC; the individual respiration rates of phytoplankton and bacteria; contributions of phytoplankton and bacteria to net production and their reliance on the mean light and DOC concentrations; the combined net production of phytoplankton and bacteria; the effect of water velocity on surface oxygen exchange and the contribution of Commonwealth environmental water to improving velocities and reducing the likelihood of low DO. These understandings are critical to predicting the likely changes in metabolism and net production within a river reach due to the delivery of environmental flows of given volume and water quality. They provide a means of assessing the effects of supplying environmental flows from different catchment sources on the basal river food resources.

2.3 Matter Transport

Background

Modification to the flow regime can alter the biogeochemistry of rivers and the adjacent floodplain system. For example, reduced flow may increase the intrusion of salt into the system and decrease the export of salt from the system. Additionally, a change in the flow regime will alter the mobilisation of nutrients from the floodplain and change the primary productivity with the river. As there is continual deposition of salt onto the landscape predominately from rainfall, it will accumulate unless transported by flow and exported from the system. Environmental flows can be used to reinstate some of the natural processes, or increase the magnitude of the processes that control the availability and transport of dissolved and particulate matter. Salinity, dissolved and particulate organic nutrients, and chlorophyll *a* are often measured or modelled to understand the influence of flows on the concentrations and transport of matter.

Nutrients drive system productivity and so understanding how they are transported between the various components of riverine ecosystems can offer insights into river and estuary productivity. Dissolved inorganic nutrients are essential resources for the growth and survival of biota and are readily assimilated (Poff *et al.* 1997). Nitrogen, phosphorus and silica are particularly important because they often control the productivity of aquatic ecosystems. Flow results in the mobilisation and transport of dissolved nutrients through the leaching of nutrients from dried sediments and dead organic matter.

Particulate organic nutrients (phosphorus and nitrogen) are those nutrients incorporated into the tissue of living and dead organisms. Flow can influence particulate organic nutrient concentrations and transport through a number of mechanisms, including through increased productivity associated with elevated dissolved nutrient concentrations.

Chlorophyll *a* is a measure of phytoplankton biomass and indicative of the amount of primary production in riverine ecosystems. Flow can influence chlorophyll *a* concentrations and transport through increased phytoplankton productivity.

To assess the contribution of environmental water delivery to matter transport in the LMR from 1 July 2014 to 30 June 2019, a hydrodynamic-biogeochemical model was applied for the region below Lock 1 to the Murray Mouth (Figure 1). The model was validated with water quality data.

Major hypotheses

Commonwealth environmental water will increase:

- The mobilisation of salt from the Basin and increase the transport of salt passing from Lock 1 through the Murray River Channel (and through the Lower Lakes and Murray Mouth)
- The mobilisation of nutrients from the Basin and increase nutrient loads passing from Lock 1 through the Murray River Channel (and through the Lower Lakes and Murray Mouth)

- Suspended solid loads (including phytoplankton biomass) passing from Lock 1 through the Murray River Channel (and through the Lower Lakes and Murray Mouth).

Methods

The contribution of environmental water to the transport of salt, nutrients and phytoplankton was assessed with a coupled hydrodynamic-biogeochemical model for the reach below Lock 1 to the Murray Mouth. Salt, nutrient and phytoplankton transport was predicted for three different flow scenarios: with all environmental water (i.e. the observed flow), flow without Commonwealth environmental water, and flow without any environmental water.

When modelling, it is necessary to make assumptions on the relationships between flow and nutrients or salt, nutrient dynamics in sediments and floodplain habitats, and the utilisation of nutrients by phytoplankton. This leads to a degree of uncertainty in model outputs; however, it is considered that this uncertainty is within reasonable bounds (Aldridge *et al.* 2013) and the results can be used to assess the general response to environmental water.

Water quality sampling and analyses

Water quality was monitored for the Murray River Channel (at Wellington), Lower Lakes and Coorong between 1 July 2014 and 30 June 2016, and for the Murray River Channel (at Morgan) between 1 July 2013 and 30 June 2019 (Table 7). At each sampling site, measurements of water temperature, electrical conductivity, dissolved oxygen, pH and turbidity were taken. In addition, integrated-depth water samples were collected and sent to the Australian Water Quality Centre, a National Association of Testing Authorities accredited laboratory. Samples were analysed for filterable reactive phosphorus (hereafter referred to as phosphate), total phosphorus (TP), combined nitrate and nitrite (NOx), ammonium, total Kjeldahl nitrogen (TN), dissolved silica and chlorophyll *a* using standard techniques. Organic nitrogen was calculated as the difference between total Kjeldahl nitrogen and ammonium.

Table 7. Water quality sampling sites within each water-body.

Water-body	Sampling site	Sampling frequency	Data source
Murray River Channel	Morgan	Approximately weekly between 01/07/2013 and 30/06/19	SA Water
	Wellington	Approximately four times between 01/07/2014 and 30/06/16	Murray Futures (DEW)
Lower Lakes	Lake Alexandrina Opening		
	Poltalloch		
	Milang		
	Lake Alexandrina Middle		
	Point McLeay		
	Finniss River		
	Currency Creek		
	Goolwa Barrage		
	Lake Albert Opening		

Water-body	Sampling site	Sampling frequency	Data source
Lower Lakes	Lake Albert Middle	Approximately four times between 01/07/2014 and 30/06/16	Murray Futures (DEW)
	Meningie		
Coorong	Monument Road		
	Murray Mouth		
	Ewe Island		
	Tauwitchere		
	Mark Point		
	Long Point		
	Parnka Point		
	Villa de Yumpa		
	Jack Point (north)		
	Salt Creek (south)		

Hydrodynamic–biogeochemical modelling

The model platform used to assess the effects of environmental water delivery on salt and nutrient transport was the coupled hydrodynamic-biogeochemical model TUFLOW-FV-AED, developed by BMTWBM and the University of Western Australia. TUFLOW-FV is now used extensively in the region for hydrological purposes, and was used to assess the contribution of environmental water to dissolved and particulate matter for water years 2013-14 to 2017-18 (Ye *et al.* 2016a; 2016b; 2017; 2018; 2019). A single model domain was applied spanning Lock 1 to the Southern Ocean, including the Coorong (Table 7; Figure 21). The TUFLOW-FV model (BMTWBM) adopts an unstructured-grid model that simulates velocity, temperature and salinity dynamics in response to meteorological and inflow dynamics. In this application, the Aquatic EcoDynamics (AED) model was configured to simulate the dynamics of light, oxygen, nutrients, organic matter, turbidity and phytoplankton.

The model runs were initialised with data from a range of data sources. Inflow data (Lock 1), used to drive the main river domain, were provided by the MDBA for the three scenarios (Figure 22), i.e. with all environmental water (“with all water”, representing observed conditions), without Commonwealth environmental water (“no CEW”), and without any environmental water (“no eWater”). These simulations were run for the period between 1 July 2014 and 30 June 2019.

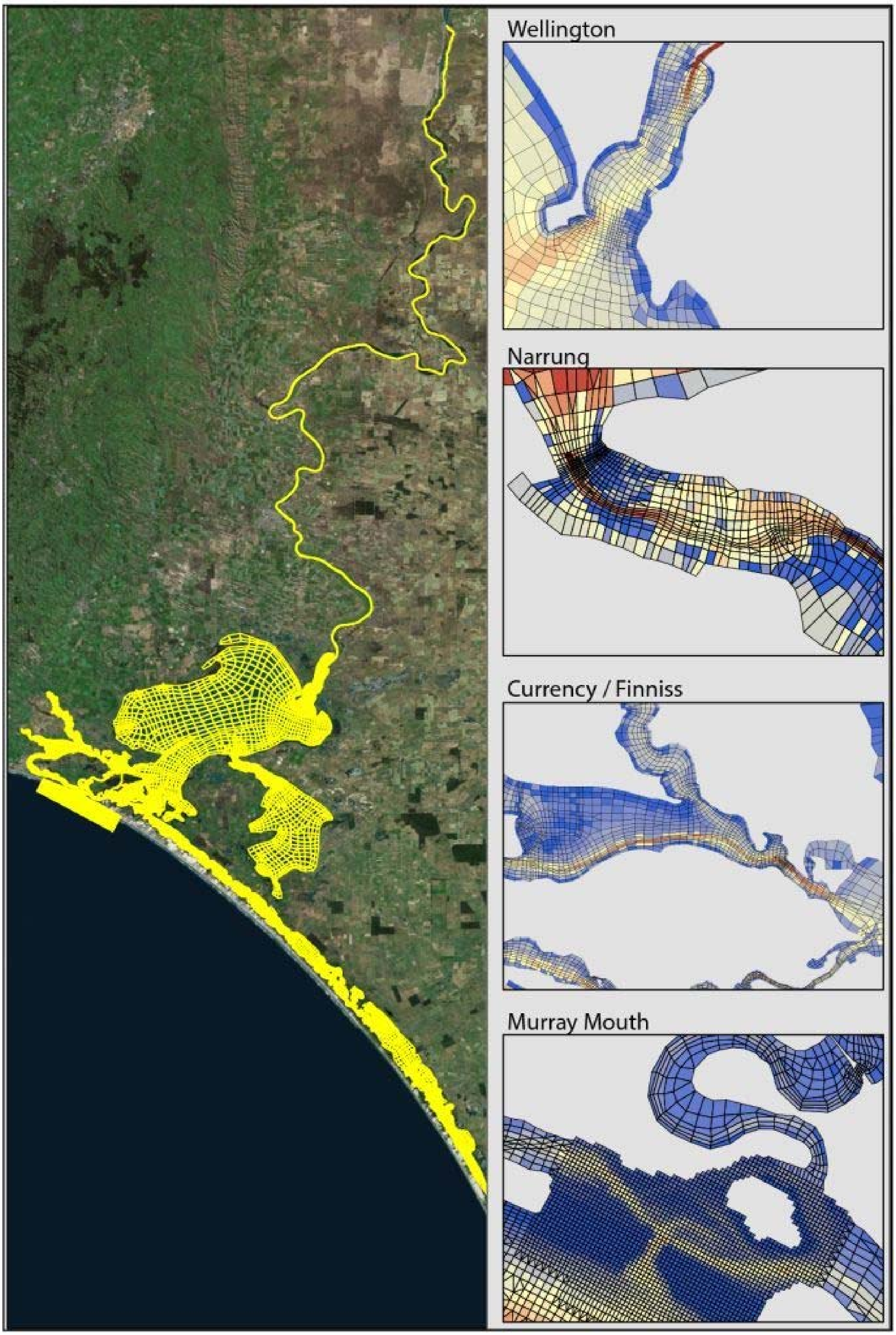

Figure 21. Overview of model domain applied in the Matter Transport study of LTIM using TUFLOW-FV. Grid provided courtesy of DEW. Coloured grids in maps on the right-hand side represent depths, i.e. increasing depth from shallow (blue) to deep (red).

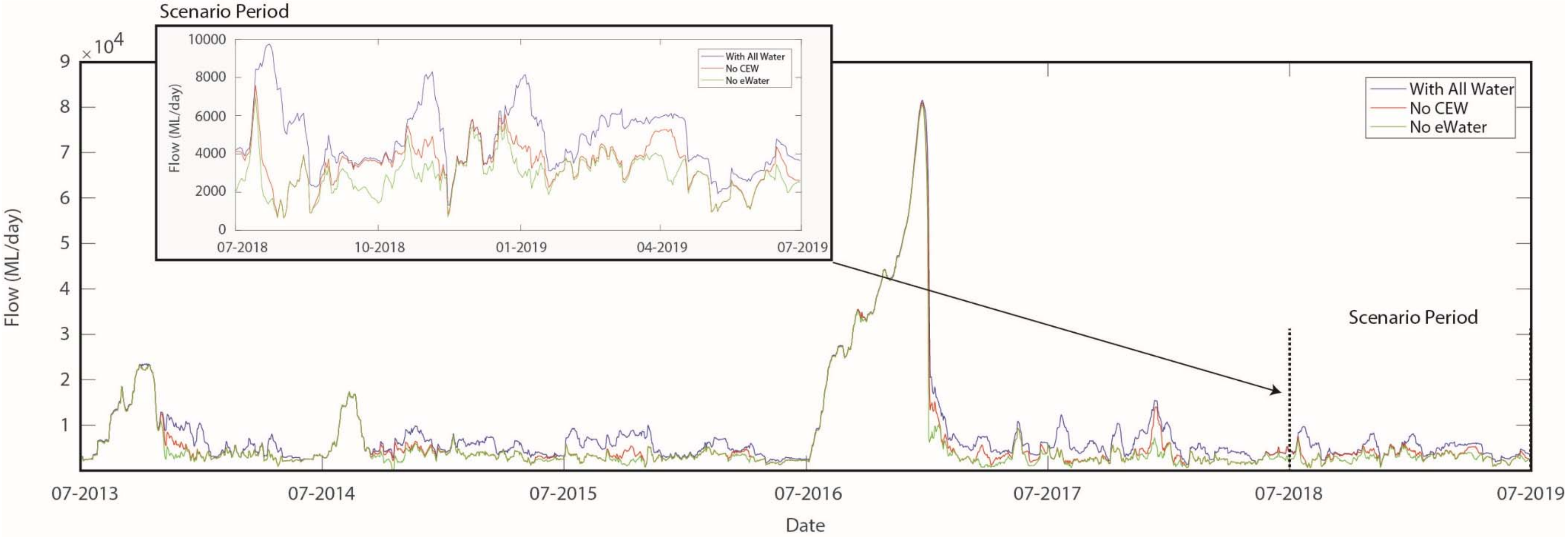

Figure 22. Overview of the three flow scenarios assessed by the model simulations. Scenarios include flow with all water, flow without Commonwealth environmental water (no CEW) and flow without any environmental water (no eWater). Flows were applied to the matter transport model at the upstream boundary, which is at Lock 1.

Additional flow specifications for SA Water off-takes were also included. Irrigation return flows were assumed to be negligible over this period and were not included in the model. Similarly, flows from Eastern Mount Lofty Ranges were not included since their contribution to the Lower Lakes is considered to be relatively minor (Cook *et al.* 2010). Meteorological conditions were based on data from Narrung. Between Lake Alexandrina and the Coorong, five barrages were included (Goolwa, Mundoo, Boundary Creek, Ewe Island and Tauwitchere) and set with a spill-over height of 0.72 m AHD. The barrage operation was set to include gate operation based on operational information provided through discussions with representatives of DEW. At the bottom of the domain, two open boundaries were specified, one at the Murray Mouth and one at Salt Creek. Murray Mouth water level was based on Victor Harbor tidal data, which are available at 10 minute resolution. Salt Creek flow data were set based on available flow data from the WaterConnect website (DEW).

Water quality conditions for both boundary points were set based on a linear interpolation of the measured nutrient and salinity data collected as part of this study. Water quality conditions for the river inflow at Lock 1 were determined based on interpolation of available data from Lock 1 or Morgan. For water quality properties for the without environmental water scenarios, rating curves were developed for flow and concentration. Based on the daily flow difference, a scaled concentration was estimated for water quality parameters including salinity, phosphate, ammonium, nitrate, total nitrogen and silica. The physico-chemical information at other sites was used to validate the model.

The influence of environmental water on the concentrations of matter was assessed through a comparison of modelled concentrations for the various scenarios for the Murray River Channel (Wellington), Lower Lakes (Lake Alexandrina Middle) and Coorong (Murray Mouth). Modelled concentrations are presented as medians of modelled cells within areas surrounding sampling sites (Figure 23). A range in concentrations within those cells is also presented for the ‘with all water’ scenario.

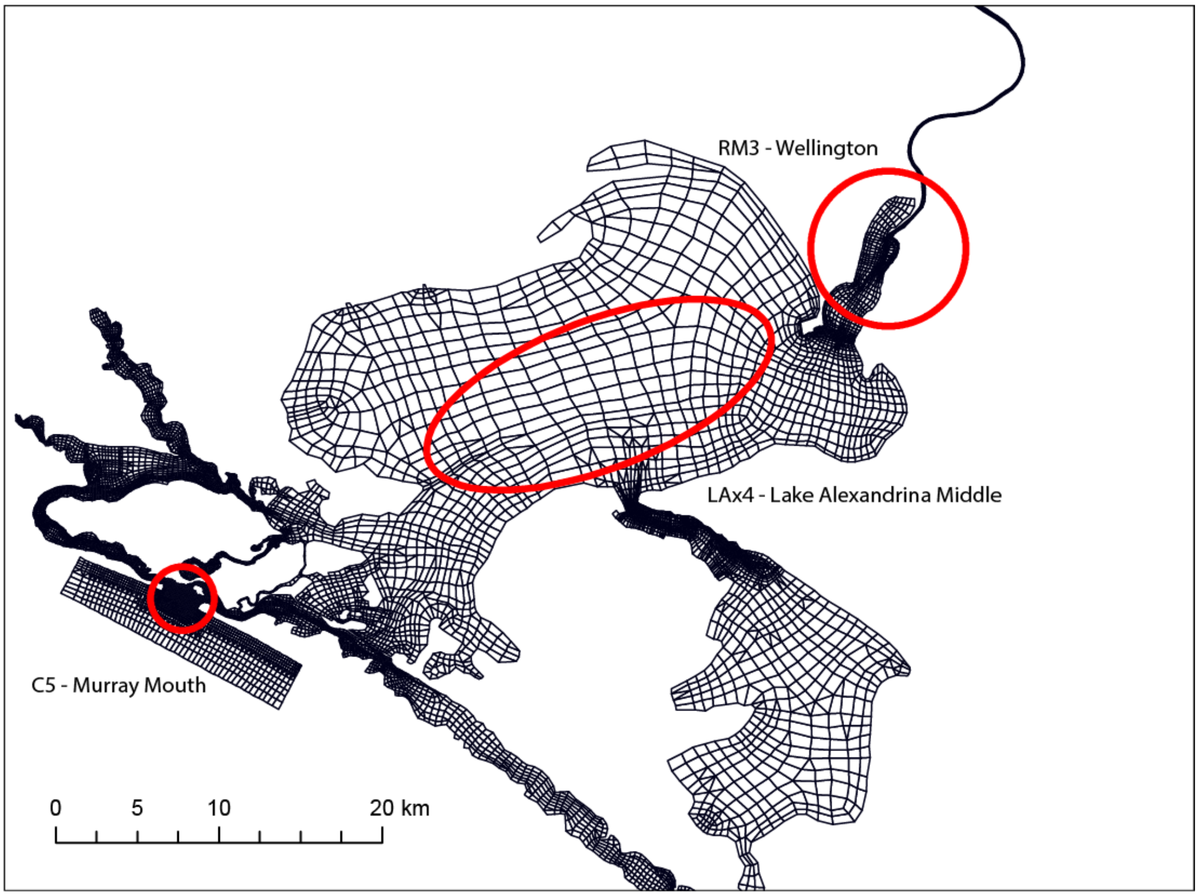

Figure 23. Modelled cells (circled) used for calculating the modelled concentration of nutrients or salt at the Wellington, Lake Alexandrina Middle and Murray Mouth sites.

The transport of matter was assessed through modelled exports from the Murray River Channel (Wellington), Lower Lakes (Barrages) and Coorong (Murray Mouth). Findings are presented for salinity, ammonium, phosphate, dissolved silica, organic nitrogen, organic phosphorus and chlorophyll *a*. Salinity is presented as practical salinity units (PSU), a measurement of the measured conductivity to standard potassium chloride (KCl) conductivity. PSU was used for validating model outputs as it overcomes observed differences in electrical conductivity caused by changes in water temperature. One PSU is approximately equal to one part per thousand.

The inflow data that were used to drive the main river domain are treated as indicative only as they do not account for all complexities associated with water accounting, water attenuation through the system and different management decisions that may have been made if the volume of environmental water provided had not been available (Neville Garland, MDBA, pers. comm.). Assumptions made to address these complexities result in uncertainty in the model outputs and so outputs are not to be treated as absolute values (refer to Aldridge *et al.* 2013 for more detail). When assessing the relative differences between scenarios, the uncertainties are considered to influence the accuracy of each scenario equally and so the model outputs are used to assess the general response to environmental water delivery.

Results

Salinity

Environmental water reduced salt concentrations in the Murray River Channel (Wellington), Lake Alexandrina and the Coorong at the Murray Mouth during each of the five years of LTIM monitoring (Figure 24). In 2017-18, for example, the median salinity in the Coorong at the Murray Mouth was 26.20 PSU across the entire year. Without Commonwealth environmental water, it would have been approximately seawater salinity (33.85 PSU).

Environmental water increased salt export over the barrages over the five-year period (Figure 24; Table 8). During the low flow years, Commonwealth environmental water accounted for 64, 87, 69 and 70% of salt export over the barrages, respectively, in 2014-15, 2015-16, 2017-18 and 2018-19. In the high flow year (2016-17), a total of 1.5 million tonnes of salt was exported over the barrages and Commonwealth environmental water contributed 8% (120,867 tonnes) (Table 8).

Like all other low flow years, in 2018-19, there was a net import of salt into the Coorong (through the Murray Mouth) of 2.68 million tonnes (Table 8). Without environmental water, the net import of salt would have been 5.44 million tonnes. Environmental water decreased salt import by 2.76 million tonnes, of which 2.47 million tonnes was attributable to Commonwealth environmental water. In 2016-17, 3.68 million tonnes of salt was exported from the Coorong, of which 0.52 million tonnes was attributable to Commonwealth environmental water.

Table 8. Five year record of modelled salt export (tonnes) over the barrages to the Coorong estuary and through the Murray Mouth into the Southern Ocean.

Scenario	**2014-15**	**2015-16**	**2016-17**	**2017-18**	**2018-19**
Barrages					
With all water	446,855	288,516	1,504,541	349,893	228,293
No CEW	161,791	36,884	1,383,674	109,171	67,396
No eWater	152,406	31,031	1,317,791	48,923	0
Murray Mouth					
With all water	-157,852	-1,850,028	3,679,277	-527,042	-2,680,574
No CEW	-3,202,552	-6,441,297	3,159,985	-3,459,211	-5,151,627
No eWater	-5,048,511	-6,649,380	1,958,989	-6,115,353	-5,438,075

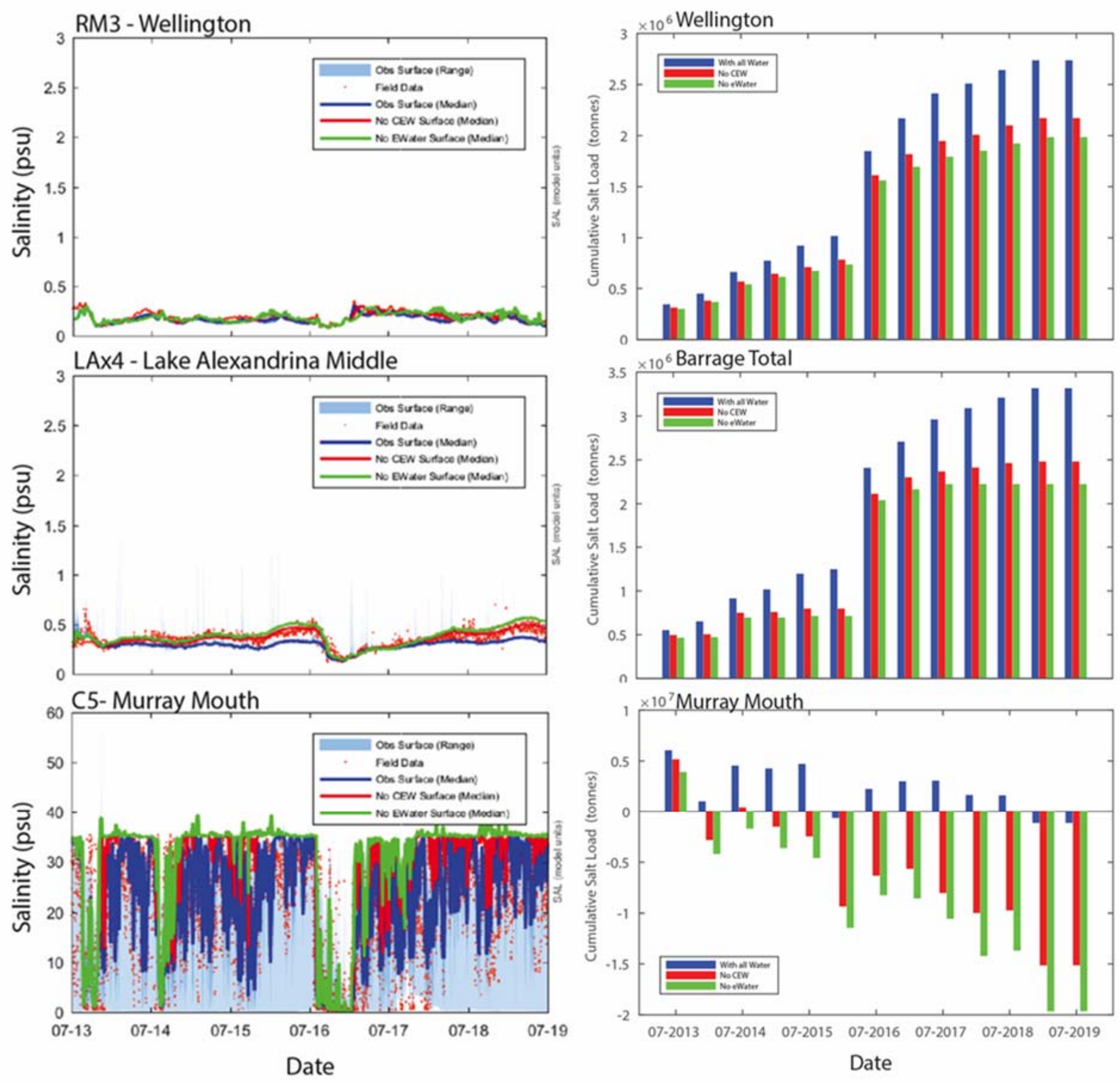

Figure 24. Modelled daily salinity concentrations (left) and six monthly cumulative salt exports (net, right) with and without environmental water delivery for 2013–2019. Scenarios include with all water, without Commonwealth environmental water (no CEW) and without any environmental water (no eWater).

Dissolved nutrients

The median concentrations of nitrogen and phosphorus do not vary considerably for any of the three modelled scenarios in any of the years between 2014–2019. Phosphate in Lake Alexandrina showed the greatest proportional difference, however, the difference between phosphate with no environmental water and with all water was only 6 µg/L in 2018-19 (Table D1 in Appendix D). Environmental water contributed considerably to the transport of nutrients, but this was primarily due to additional flow not a change in the nutrient concentrations. In low flow years, total nitrogen (TN) export ranged between 1,238–2,174 tonnes and total phosphorus (TP) export ranged between 113–204 tonnes. In the high flow year, the TN and TP export were 11,029 and 1,049 tonnes, respectively. The annual TN export attributable to Commonwealth environmental water ranged 609–1,507.8 tonnes, and TP export attributable to Commonwealth environmental water ranged 11.1–137 tonnes. In 2018-19, the transport of phosphate over the barrages would have been nil without environmental water (no e-water) and 0.113 tonnes without Commonwealth environmental water.

The silica load over the barrages ranged 9,428–14,587 tonnes in low flow years, considerably lower than the 70,207 tonnes that flowed over the barrages in 2016-17. Silica export attributable to Commonwealth environmental water was 0–6,836 tonnes.

Chlorophyll a

Commonwealth environmental water contribution towards chlorophyll transport over the barrages to the Murray Mouth varied across years (Table 9). In 2015-16, Commonwealth environmental water contributed to 92% (22 tonnes) of the total export of phytoplankton biomass. During high flows in 2016-17, the total phytoplankton loads were two orders of magnitude higher than 2015-16. In 2016-17, Commonwealth environmental water contributed to 6% (191 tonnes) of the total export of phytoplankton biomass over the barrages.

Table 9. Five year record of modelled phytoplankton export (as carbon, tonnes) over the barrages to the Coorong estuary.

Scenario	2014-15	2015-16	2016-17	2017-18	2018-19
With all water	27	23.5	2,939	221.5	5.8
No CEW	21	1.9	2,748	63.9	1.6
No eWater	19	1.5	2,629	25.7	0

Evaluation

The evaluation approach, including assessment criteria, is described in the evaluation section for hydrological regime (Section 2.1).

Table 10. Matter Transport evaluation questions and answers. CEW = Commonwealth environmental water.

CEWO evaluation questions		Outcomes of CEW delivery				
		2014-15	2015-16	2016-17	2017-18	2018-19
What did CEW contribute to salinity:	Levels? Reduction at the Murray Mouth due to CEW (Median salinity presented, PSU)	From 34.6 to 24.0	From 35.1 to 29.7	From 21.6 to 11.3	From 34.4 to 26.2	From 34.8 (without CEW) to 30.9 (with CEW)
	Transport? Additional export over barrages due to CEW (tonnes salt per year)	285,064	251,632	120,867	240,722	160,897
What did CEW contribute to the salinity regime?		CEW reduced salinity in the Murray River Channel and Lower Lakes; increased salt export over the barrages; and reduced salt intrusion into the Murray Mouth from the ocean, which reduced salinity in the Coorong. CEW has played a key role in salt export from the Basin, accounting for 64, 87, 69 and 70% of salt export, during the four years of low flow (2014-15, 2015-16, 2017-18 and 2018-19), respectively. In the low flow years, the total salt export ranged 228,293–446,855 tonnes, which is well below the Basin Plan target of 2 million tonnes of salt per year. In the high flow year (2016-17), 1.5 million tonnes was exported and CEW contributed 8% (120,867 tonnes) of salt export.				

CEWO evaluation questions		**Outcomes of CEW delivery**				
		2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to nutrient concentrations and transport?	Nitrogen concentrations? Reduction in Lake Alexandrina due to CEW (median TN, mg/L)	1.77 to 1.57	2.05 to 1.58	1.94 to 1.67	2.33 to 1.85	2.78 to 1.95
		CEW played a negligible/minor role in reducing nitrogen concentrations from 2014–2019. It is not determined whether the levels of change are biologically important.				
	Transport? Export over barrages due to CEW (tonnes)	609	1,007	123	1,508	816
		CEW increased nitrogen export as nitrogen load was largely a function of flow volume.				
	Phosphorus concentrations? Reduction in Lake Alexandrina due to CEW (median TP, mg/L)	0.166 to 0.146	0.192 to 0.144	0.182 to 0.162	0.224 to 0.174	0.282 to 0.194
		CEW played a minor role in altering phosphorus concentrations from 2014–2019. Lake Alexander normally acts as a sink for phosphorus. As the nutrient concentrations are reasonably high, a reduction acts to reduce the risk of problematic algal blooms.				
	Transport? Export over barrages due to CEW (tonnes)	54	90	11	137	77
		CEW increased phosphorus export as phosphate load was largely a function of flow volume.				

CEWO evaluation questions		**Outcomes of CEW delivery**				
		2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to nutrient concentrations and transport?	Silica concentrations? Reduction in Lake Alexandrina due to CEW (mg/L)	11.5 to 9.52	19.7 to 11.4	15.3 to 12.5	17.8 to 11.6	24.8 to 15.1
		CEW played a negligible/minor role in altering silica concentrations but it is not determined whether the levels of change in silica are biologically important.				
	Transport? Export over barrages due to CEW (tonnes)	3,551	6,836	0	8,787	5,469
		CEW increased silica export as silica load was largely a function of flow volume. 2016-17 was a flood year and so the silica export that year may be a function of the concentrations used to model. Total silica export was 70,207 tonnes, although not attributed to CEW.				
What did CEW contribute to phytoplankton:	Concentrations? Change in Lake Alexandrina due to CEW (mg/L)	12.5 to 13.8	9.50 to 10.7	23.9 to 23.9	15.4 to 14.2	10.8 to 9.02
		CEW played a negligible role in altering chlorophyll concentrations from 2014–2019. At different flows, different phytoplankton species will dominate; higher flow conditions will reduce the risk of cyanobacteria blooms and promote the diatom *Aulacoseira*, which are a nutritious food source to support the riverine food web.				
	Transport? Export over barrages due to CEW (tonnes)	8	6	14	12	6
		CEW increased phytoplankton export as chlorophyll load was a function of flow volume.				
What did CEW contribute to water quality to support aquatic biota and normal biogeochemical processes?		CEW had substantial contribution to salinity reduction and maintaining estuarine habitats, and so would have promoted biodiversity in the Coorong. CEW also aided in lowering salinity in the Lower Lakes which would have benefits for the freshwater biota.				

CEWO evaluation questions	Outcomes of CEW delivery				
	2014-15	2015-16	2016-17	2017-18	2018-19
What did CEW contribute to ecosystem function? (LMR, Lakes)	CEW exported additional 1.06 million tonnes of salt out of the Basin (through the Murray barrages) over the last five years, leading to reduced salinity in the Murray River Channel and Lower Lakes. This is important in maintaining aquatic habitat quality and supporting freshwater biodiversity. CEW delivery increased exchange of nutrients and phytoplankton between critical habitats of the LMR, which may have supported primary and secondary productivity in the region and in doing so, supported food webs of the Murray River Channel and Lower Lakes.				
What did CEW contribute to ecosystem function? (Coorong)	The decrease in salt that entered the Murray Mouth with CEW deliveries meant that not as much salt accumulated in the Coorong. Maintaining appropriate salinity levels ensure the favourable conditions for ecosystem and life-history processes of biota in the Coorong. Without environmental water, there would have been an additional 20 million tonnes of salt enter the Coorong over the period 2014–2019. This would have produced salinity in the South Lagoon reminiscent of the end of the Millennium drought. At this time the salinity was five times seawater and there was detrimental loss of ***Ruppia tuberosa***, fish and waterbirds. CEW delivery via barrage releases increased nutrients and phytoplankton input, which likely enhanced primary and secondary productivity and supported food webs in the Murray estuary, Coorong and adjacent marine ecosystems.				

Contribution (to what extent CEW contributed towards the outcome, with the significance of the outcome considered):
⬜ Unknown 🟥 Negative ⬜ None/negligible 🟩 Minor 🟩 Moderate 🟦 Substantial

Discussion

Salinity

Environmental water has diluted salt in the LMR channel (Wellington), Lake Alexandrina and the Coorong (the Murray Mouth). The salinity was maintained well within the range required for potable water in the river and lake over the last five years, but water was about 10% fresher with the environmental flows. The median salinity in the Murray Mouth in 2018-19 was 30.9 PSU, which was similar to 2017-18 (median salinity 26.20 PSU) but higher than in 2016-17 (12.97 PSU), where flow into South Australia peaked at 94,600 ML/d.

Nevertheless, Commonwealth environmental water created fresher conditions at the Murray Mouth in 2018-19, compared to without environmental water.

Salinity in the Coorong is primarily a function of riverine inflows and tidal movement. When barrage flows are low, seawater enters the Murray Mouth and salt can accumulate in the Coorong. The relatively low flows in 2014–2017 and 2018-19 meant that there was considerable import of salt (i.e. 527,042 tonnes in 2017-18 and 2.98 million tonnes in 2018-19) from the sea to the Coorong. Environmental water played a key role in reducing salt import into the Coorong. Without environmental water, the net import of salt would have been 6.1 million tonnes in 2017-18 and 5.4 million tonnes in 2018-19. During the Millennium Drought, and particularly in 2008 and 2009, the import of salt into the Coorong resulted in salinity in the South Lagoon that was five times seawater salinity, and demise of much of the aquatic life (Brookes *et al.* 2009). Environmental water provides freshening flows but also acts to inhibit seawater intrusions, thereby maintaining more appropriate salinity conditions in the Coorong. Without the environmental water there would have been an additional 20 million tonnes of salt enter the Coorong over the period 2014–2019.

Dissolved Nutrients

Environmental water contributed to the transport of nutrients, but this was primarily due to additional volume not a change in the nutrient concentrations. It was evident that environmental flows contributed a considerable load of nutrients to the Murray Mouth. From this evidence, it can be concluded that over the last 5 years environmental flows were a key driver in promoting estuarine productivity. The particulate nutrient load was much higher than the dissolved fractions. Environmental water was responsible for a large proportion of the particulate organic nutrient load over the barrages with Commonwealth environmental water accounting for approximately 67% in 2018-19. If there was no environmental water, the nutrient load over the barrages would have been zero.

Chlorophyll a

Chlorophyll *a* is a photosynthetic pigment that is ubiquitous in the phytoplankton, so is often used as a measure of the relative size of the phytoplankton community. A considerable amount of the total organic nutrients is likely to be bound within phytoplankton, and so the chlorophyll loads reflect the loads of particulate organic nitrogen and phosphorus. Chlorophyll export can be interpreted as a transfer of food resources from one site to another. As flow in 2017-18 and 2018-19 was low, there was limited transfer of food from the river to the Coorong and coastal environments. During the high flow of 2016-17, the river discharge likely played a significant role in promoting estuarine productivity, contributing 167 tonnes of chlorophyll over the barrages to the Murray Mouth.

Management implications

There is approximately 10^{11} tonnes of salt in groundwater in the MDB and an additional 1.5 million tonnes of salt is deposited in the basin each year by rainfall (Herczeg *et al.* 2001). Unless salt is exported from the basin with flow, there will be a net accumulation of salt within the basin. The Basin Plan sets out a salt export objective (section 9.09) to ensure adequate flushing of salt from the Murray River system into the Southern Ocean. The Basin Plan's indicative target for salt export from the basin is 2 million tonnes per year. The five years of salt export modelling enable the contribution of environmental flows to salt export to be scrutinised (Table 8). Flow has been relatively low in four of the five years of LTIM monitoring. In the low flow years (2014-15, 2015-16, 2017-18 and 2018-19) Commonwealth environmental water plays a key role in salt export from the basin, accounting for 64–87% of total salt export. In the high flow year (2016-17), 1.5 million tonnes was exported and Commonwealth environmental water contributed 8%.

Maximum exports of matter from the Murray Mouth are likely to be achieved by delivering environmental water during periods of low oceanic water levels (e.g. summer). In contrast, environmental water delivery to the Murray River Channel at times of high oceanic water levels is likely to increase the exchange of water and associated nutrients and salt through the Coorong, rather than predominately through the Murray Mouth. This may decrease salinities and increase productivity within the Coorong more than what would occur if water is delivered at times of low oceanic water levels.

The load of nutrients exported from the basin over the barrages is an interesting issue; on one hand nutrient export drives estuarine productivity, but on the other hand it is desirable to maintain an appropriate level of nutrients in the catchment where they can support aquatic productivity. It is not possible, with the current understanding, to make a judgement as to what is a desirable load to balance the need between systems. In the last five years, the dissolved nutrient concentrations were relatively low and the chlorophyll concentrations in the river, lake and Murray Mouth sites indicated the system was mesotrophic, i.e. had intermediate levels of productivity.

Conclusion

The contributions of environmental water appear to have significantly increased the exchange of dissolved and particulate matter through the LMR to the Southern Ocean. In low flow years, environmental flow delivery can play a key role in salt export from the Basin. In previous low flow years (2014-15, 2015-16 and 2017-18), environmental water has contributed 66–89% of salt export. In 2018-19, however, environmental flow was responsible for all salt export. Environmental water is also critical in reducing salt import into the Murray Mouth from the ocean, lowering salinity in the Coorong and maintaining estuarine habitat to support ecological functions and biodiversity. Environmental flow deliveries during periods when there would otherwise be negligible water exchange between the Lower Lakes and Coorong can promote connectivity and allow matter exchange between these two water-bodies.

2.4 Microinvertebrates

Background

Aquatic microinvertebrates (microcrustaceans, rotifers and protists) are critical components of aquatic food chains. They are consumers of bacteria, algae and other microinvertebrates, and a major food source for larger organisms (e.g. macroinvertebrates) (Schmid-Araya and Schmid 2000; Pernthaler and Posch 2009). At the top end of food chains they are important first-feeds for early life stages of fish (i.e. larvae) (Arumugam and Geddes 1987; Tonkin *et al.* 2006). For example, availability of suitable microinvertebrate prey (e.g. copepods and cladocerans) for fish during the switch from endogenous (yolk sac absorption) to exogenous feeding can determine their survival and the level of recruitment success (year-class strength).

Aquatic microinvertebrates are rapid responders to environmental flows, as some organisms are often imported with the incoming water (hereafter termed the 'imported community') (Jenkins and Boulton 2003), while others emerge from diapause eggs within the sediment (Boulton and Lloyd 1992) and change the species composition and diversity of the resident assemblage within days (Tan and Shiel 1993). Different habitat types promote different assemblages of microinvertebrates within riverine ecosystems. At a very simplistic level, microinvertebrate assemblages can be categorised as littoral, limnetic, lotic or benthic. Following inundation of habitat, only a fraction of the organisms within the egg-bank emerges and combines with the imported community. This fraction is the 'active egg-bank' (as defined by Caceres and Hairston 1998). The remaining eggs supplement the 'persistent egg-bank' (see Brendonck and De Meester 2003), which provides an important buffer against periods when the active egg-bank becomes depleted. Due to this tendency for eggs to accumulate, the egg-bank can store a diverse array of organisms from past inundation events, which play an important role in future populations and communities.

One of the most important factors that influences the microinvertebrate community across all habits is water residence time (WRT). Generally, WRT has a strong positive relationship with microinvertebrate density and biomass. High microinvertebrate density is favoured by increased WRT due to their food resources often being more available and because microinvertebrates cannot reproduce, maintain their position, or depending on the species, survive within fast flowing water (e.g. Richardson 1992; Sluss *et al.* 2008). Therefore, most species depend on still or slow flowing water environments to maintain their populations. Consequently, longer WRT's favour microinvertebrate development within off-channel sites including the littoral and limnetic habitats of floodplain lakes, billabongs and wetlands. Within in-channel sites, microinvertebrate community development occurs predominantly in weir pools and in littoral and limnetic backwater habitats (e.g. Reckendorfer *et al.* 1999; Baranyi *et al.* 2002). Organisms can then be transferred between habitats by hydrological mixing and exchange, or due to the entrainment of organisms from limnetic or littoral into lotic habitats. This is however, a very simplistic overview, with successional changes occurring and the physico-chemical environment also playing a major role in driving community dynamics.

Major hypotheses

- Microinvertebrate species richness will increase with discharge (influenced by environmental water) due to organisms emerging from the egg bank and the entrainment of organisms from a wider range of habitats in the main river channel.
- Microinvertebrate density will increase with discharge (influenced by environmental water) and temperature due to the inundation and flushing of slow-flowing habitats and increased productivity.
- During times of longitudinal connectivity (influenced by environmental water), the microinvertebrate assemblage structure in the LMR will reflect the community assemblage of its source community (e.g. during times in which discharge is coming from the Darling River, tropical species will be present in the LMR).
- Microinvertebrate assemblage responses to discharge (including environmental water) will be reflected in the dietary components of fish larvae (golden perch).

Methods

Sampling sites and procedure

Microinvertebrate sampling was conducted approximately fortnightly between September and January each year at three core LTIM sites within each of the floodplain and gorge geomorphic zones of the LMR (Table 11 and Table 12), concurrent with larval fish sampling (Section 2.5). Three replicate samples were taken at each site during the day.

Table 11. Microinvertebrate sampling dates from 2014–2018 in the LMR.

Trip no.	2014-15	2015-16	2016-17	2017-18
1			26-28/09/2016	
2		6-7/10/2015	11-12/10/2016	3-4/10/2017
3		20-21/10/2015	24-25/10/2016	16-17/10/2017
4	3-4/11/2014	2-3/11/2015	7-8/11/2016	30-31/10/2017
5	19-20/11/2014	17-18/11/2015	21-22/11/2016	13-14/11/2017
6	1-2/12/2014	30/11-1/12/2015	6-8/12/2016	27-28/11/2017
7	14-15/12/2014	15-16/12/2015	21/12/2016	11-12/12/2017
8	7-8/01/2015	5-6/01/2016	10-11/01/2017	3-4/01/2018
9	19-20/01/2015	20-21/01/2016		

A Perspex Haney plankton trap (4.5 L capacity) was used mid-channel (by boat) to collect surface and bottom volumes (9 L), which were filtered through a 37 µm-mesh plankton net suspended in a bucket and rinsed into a 200 ml PET bottle screwed to a purpose-built ferrule at the net end. The filtrate was then preserved in the field (100% ethanol) to a final concentration of ~75%, and a volume of <200 ml. In the laboratory, the sample was decanted into a measuring cylinder, the volume noted, the cylinder agitated, and a 1 ml aliquot withdrawn using a Gilson auto pipette. This 1 ml was run into a Pyrex 1 ml Sedgewick-Rafter cell, and the microinvertebrates present were counted and identified. Counts for each sample were based on a single subsample.

Table 12. Details of microinvertebrate sampling sites downstream (DS) of Lock 1 and Lock 6 in the LMR.

Zone	Site	Latitude	Longitude
Floodplain	5 km DS Lock 6	S34.01902	E140.87572
Floodplain	7 km DS Lock 6	S34.01764	E140.85461
Floodplain	9 km DS Lock 6	S34.0319	E139.61723
Gorge	5 km DS Lock 1	S34.4052	E139.61723
Gorge	7 km DS Lock 1	S34.42263	E139.61293
Gorge	9 km DS Lock 1	S34.44596	E139.61102

Statistical analyses

To assess the influence of Commonwealth environmental water on microinvertebrate density, species richness and assemblage structure (including rotifers, cladocerans and copepods) over the long-term (4 years), variation between sampling years (i.e. 2014-15, 2015-16, 2016-17 and 2017-18) and sites was investigated. Protists and macroinvertebrates were excluded to allow multi-year analysis. Temporal and spatial variation in microinvertebrate mean density, mean species richness and assemblage structure was analysed using permutational multivariate analysis of variance (PERMANOVA), Similarity Percentages (SIMPER) analysis and Non-metric Multi-Dimensional Scaling (MDS) in the software package PRIMER v. 6.1.12 (Clarke and Gorley 2006) and PERMANOVA + v.1.02 (Anderson *et al.* 2008). To simplify all analysis, the three replicates from each of the three sites within both geomorphic zones were averaged, and the average from each site used as a replicate for the corresponding zone (e.g. the average of site one below Lock 6 was used as a single replicate for the floodplain geomorphic zone). Trips 4 to 8 (between early November and January) were the only trips that were consistent between all four years and therefore other trips were excluded from PERMANOVA analysis for inter-annual comparisons. Analysis was conducted on each year independently and included all trips for intra-annual comparisons. In instances when multiple comparison were made, the Bonferroni correction was not applied due to its conservative nature and potential for increasing type II error. Therefore, raw alpha values have been presented and all post-hoc statistically significant results interpreted within the context and with careful consideration of current conceptual understanding and other research results. A 50% contribution cut off was applied to SIMPER analysis. Species assemblage data were graphically presented in MDS plots. Species density data was averaged across sites to reduce overcrowding within the MDS plots. Two separate MDS plots were generated, one with species correlations and a second with water temperature and discharge correlations overlain. The two plots were combined to create a single plot with both sets of correlations. Each species was classified as either littoral, littoral (facultatively pelagic) or pelagic and used to simplify labelling. For definitions of these categories and species categorisation, see Appendix D. All statistical analyses were conducted on square-root transformed day-time Haney trap data from below Lock 1 and Lock 6 between early November and January. When the number of unique permutations were low (i.e. <100), Monte Carlo p-values have been reported. All results are reported to two significant figures and ±1 standard error reported in brackets.

Larval gut analysis

To determine if Commonwealth environmental water contributed to the timing of microinvertebrate productivity and presence of key species in relation to diet of large-bodied fish larvae, diet composition of fish larvae was assessed. Gut contents of golden perch, freshwater catfish and Murray cod larvae, collected opportunistically through larval fish sampling as part of the Fish Spawning and Recruitment indicator (Table 13; Section 2.5), were analysed using traditional taxonomic methods. The abundance of microinvertebrate prey in the stomachs of fish larvae (by species) were summarised in stacked bar charts.

Table 13. Sample sizes of large-bodied fish larvae analysed for gut contents from 2014–2018, indicating the number of fish with contents in their stomachs. Total lengths (TL) are for all fish sampled.

	Freshwater catfish		**Golden perch**		**Murray cod**	
Year	TL (mm)	*n* food (*n* total)	TL (mm)	*n* food (*n* total)	TL (mm)	*n* food (*n* total)
2014/15	12.0–15.0	3(7)	11.0–14.0	2(2)	10.0–12.0	5(16)
2015/16	12.3–20.7	9(14)	12.9	1(1)	9.7–14.7	8(29)
2016/17		0(0)		0(0)	7.8–12.8	2(20)
2017/18	14.0–16.0	8(10)	3.5–20.0	10(13)	12.0–38.0	0(3)

Results

Inter-annual comparisons

Throughout the four years, microinvertebrate density varied between 72 (±6.09) and 2,408 (±100) ind/L (Figure 25 and Figure 26). A PERMANOVA on mean density indicated that there was a significant difference between years (*P*≤0.001) and sites (*P*≤0.001) (Table 14). There was no significant interaction between year and site (*P*=0.055). Pairwise comparisons indicated that microinvertebrate densities were significantly different in 2014-15 in comparison to 2015-16 (*P*=0.0017) and 2017-18 (*P*=0.047), and significantly different in 2016-17 in comparison to 2015-16 (*P*=0.002) and 2017-18 (*P*=0.019) (Table 14; Figure 25 and Figure 26). Pairwise comparisons also indicated that densities were significantly different at Lock 6 in comparison to at Lock 1 (*P*≤0.001) (Table 14; Figure 25 and Figure 26).

Throughout the four years, species richness varied between 4.3 (±0.51) and 34 (±1.2) taxa (Figure 27). A PERMANOVA on mean species richness indicated that there was a significant difference between years (*P*=0.0031) and between sites (*P*=0.0042) (Table 15; Figure 27). There was no significant interaction between year and site (*P*=0.065). Pairwise comparisons indicated that mean species richness was significantly greater in 2016-17 in comparison to all other years (*P*=0.039–0.0078) (Table 15; Figure 27). Pairwise comparisons also indicated that mean species richness was significantly different between Lock 6 and Lock 1 (*P*≤0.001) (Table 15; Figure 27).

A PERMANOVA on community assemblage across the four years indicated that there was a significant interaction between year and site (P≤0.001) (Table 16). Pairwise comparisons indicated that community assemblages between Lock 6 and Lock 1 were significantly different within all years (*P*=0.0001–0.0371) (Table 16; Figure 28). Pairwise comparisons indicated that community assemblage at Lock 6 was significantly different between all years (*P*≤0.001) and community assemblage at Lock 1 was significantly different between all years (*P*≤0.001) (Table 16; Figure 28). Community assemblages changed seasonally in all years, indicating changes associated with increasing water temperature (diagonally upwards from right to left in the MDS plot) (Figure 28). Community assemblages also indicated changes in relation to increasing discharge (diagonally downwards from right to left in the MDS plot) (Figure 28).

Assemblages in 2014-15 and 2015-16, when discharge was low, were more similar than when compared to other years and demonstrated overlap between the two years (Figure 28). This however, excluded assemblages in November and December 2015-16 (trips 5, 6 and 7) at Lock 1 when discharge was considerably lower than at the same time in other years and was primarily associated with lower densities of pelagic microinvertebrates (Figure 25; Figure 26; Figure 28). Assemblages in 2016-17, when discharge was high, demonstrated the least overlap with other years and showed an association with high discharge into South Australia (QSA) throughout November and December (trips 4, 5, 6 and 7 in November and December) and greater densities of a range of pelagic and littoral organisms (Figure 25; Figure 26; Figure 28). Assemblages in 2017-18, when there was a within-channel rise of greater magnitude than 2014-15 and 2015-16, demonstrated the widest variety in community structure. This included: a distinct community in October (trips 2 and 3) in comparison to all other years; a community that fell between those detected during lower flow years (2014-15 and 2015-16) and during the flood (2016-17) in late-November and early December; and similarity with communities detected in 2016-17 during the flood, from mid-December to January. Interestingly, the distinct community detected in October was largely associated with high densities of cladocerans and copepods, particularly at Lock 6 (Figure 25; Figure 26; Figure 28).

Table 14. PERMANOVA table of results for pairwise comparisons on density data between years, where 2014 = 2014-15, 2015 = 2015-16, 2016 = 2016-17 and 2017 = 2017-18, and sites where Lk1 = Lock 1 and Lk 6 = Lock 6. *P*-values presented in bold are significant comparisons.

Groups	**t**	***P* (perm)**	**Unique perms**
Years:			
2014, 2015	3.1176	**0.0017**	9940
2014, 2016	1.224	0.2267	9929
2014, 2017	2.007	**0.0469**	9933
2015, 2016	3.111	**0.002**	9929
2015, 2017	1.2507	0.2145	9938
2016, 2017	2.3583	**0.019**	9938
Sites:			
Lk1, Lk6	4.6502	**0.0001**	9958

Table 15. PERMANOVA table of results for pairwise comparisons on species richness data between years, where 2014 = 2014-15, 2015 = 2015-16, 2016 = 2016-17 and 2017 = 2017-18, and sites where Lk1 = Lock 1 and Lk 6 = Lock 6. *P*-values presented in bold are significant comparisons.

Groups	t	*P* (perm)	Unique perms
Years:			
2014, 2015	1.2876	0.2016	9926
2014, 2016	2.7371	**0.0078**	9912
2014, 2017	0.64407	0.5236	9923
2015, 2016	3.1158	**0.0042**	9936
2015, 2017	0.72906	0.4797	9925
2016, 2017	3.0018	**0.0039**	9911
Sites:			
Lk1, Lk6	2.8099	**0.0046**	9909

Table 16. PERMANOVA table of results for pairwise comparisons on community assemblage data between years, where 2014 = 2014-15, 2015 = 2015-16, 2016 = 2016-17 and 2017 = 2017-18, and sites where Lk1 = Lock 1 and Lk 6 = Lock 6. *P*-values presented in bold are significant comparisons.

Groups	t	*P* (perm)	Unique perms
Years (Lock 1):			
2014, 2015	3.0504	**0.0001**	9924
2014, 2016	3.0323	**0.0001**	9927
2014, 2017	3.5463	**0.0001**	9937
2015, 2016	3.204	**0.0001**	9931
2015, 2017	3.4245	**0.0001**	9946
2016, 2017	3.1871	**0.0001**	9925
Years (Lock 6):			
2014, 2015	2.7953	**0.0001**	9946
2014, 2016	3.2266	**0.0001**	9937
2014, 2017	2.9587	**0.0001**	9940
2015, 2016	2.8804	**0.0001**	9922
2015, 2017	2.6148	**0.0001**	9922
2016, 2017	2.8329	**0.0001**	9923
Sites (LK1, LK6):			
2014-15	1.5623	**0.0371**	9906
2015-16	2.1983	**0.0001**	9926
2016-17	1.7633	**0.0027**	9928
2017-18	2.1464	**0.0008**	9937

Intra-annual comparisons

2014-15

In 2014-15, microinvertebrate density varied between 376 (±17) and 2,076 (±117) ind/L (Figure 25; Figure 26), while species richness varied between 11.89 (± 1.66) and 25.78 (±0.22) (Figure 27). PERMANOVAs indicated that there was a significant interaction between site and trip for both density (*P*=0.0001) and species richness (*P*=0.016), which suggests that differences in microinvertebrate composition among trips were not consistent between sites and vice versa.

In November 2014, the delivery of Commonwealth environmental water increased discharge to South Australia from ~6,000 to ~10,000 ML/d (Figure 5). At Lock 6, these increases in discharge in combination with the operation of the Chowilla regulator coincided with a higher representation of littoral (facultatively pelagic) rotifers within the microinvertebrate community and the presence of standing-water pelagic microcrustaceans (Figure 25; Figure 28; Appendix B). All taxa driving significant differences between sampling trips are known from the upper Murray and southern tributaries (Table 17). These organisms contributed to significantly greater microinvertebrate densities at Lock 6 in comparison to Lock 1 on 3 November and 19 November 2014 (*P*=0.003 and 0.0496, respectively) and the highest microinvertebrate densities observed during the study for October and early November at Lock 6 (Figure 25). At Lock 6 and Lock 1, these increases in discharge coincided with significantly higher species richness on 3 November 2014 than the 19 November 2014 trip (*P*=0.0322 at Lock 6 and *P*=0.0243 at Lock 1) (Figure 27). Species richness was high for November 2014 in comparison to those recorded in early November in all other years at both sites (excluding at Lock 1 in 2015-16 during weir pool raising) (Figure 27).

In mid-January 2015, the delivery of Commonwealth environmental water, which increased discharge from ~7,000 to 10,000 ML/d (Figure 5), coincided with the highest microinvertebrate density and species richness recorded for the sampling period at Lock 6 and Lock 1 (Figure 25; Figure 26; Figure 27). Despite microinvertebrate density commonly being greatest in January at both sites, those observed at Lock 1 were the greatest measured across all four years for that site (Figure 26). Microinvertebrate density at Lock 1 gradually increased with increasing temperature throughout the sampling period (Figure 26). Species richness was significantly higher at Lock 6 and Lock 1 on 19 January 2015 in comparison to all other trips excluding 3 November 2014 and 14 December 2014 at Lock 6 (*P*=0.0004–0.0124) (Figure 27). Contributing to the high species richness and densities in January was a high representation of littoral (facultatively pelagic) rotifers within the microinvertebrate community (Figure 25; Figure 26).

2015-16

In 2015-16, microinvertebrate density varied between 72 (±6.1) and 1,955 (±205) ind/L (Figure 25; Figure 26), while species richness varied between 5.7 (±0.69) and 26 (±1.1) taxa (Figure 27). PERMANOVAs indicated that there was a significant interaction between site and trip for both density (*P*=0.0001) and species richness (*P*=0.0001), which suggests that differences in microinvertebrate composition among trips were not consistent between sites and vice versa.

From September to November 2015, environmental water (77 % Commonwealth environmental water), maintained discharge at a peak of 9,600–11,700 ML/d, where it would have otherwise been ~4,100–4,900 ML/d (Figure 5). At Lock 6, microinvertebrate density was significantly greater on 2 November 2015 in comparison to 6 October 2015 and 20 October 2015 (*P*=0.0006) and coincided with maximum water levels at the Chowilla regulator (Figure 25). Additionally, at Lock 6 there was significantly higher species richness on 2 November 2015, 17 November 2015 and 30 November 2015 than the 20 October 2015 (*P*=0.0072, 0.0042 and 0.0271, respectively) which coincided with the drawdown of the Chowilla regulator and Lock 6 (Figure 27; Appendix B). However, these increases in density and species richness were relatively small. At Lock 1, microinvertebrate density was significantly greater on 6 October and 20 October 2015 than Lock 6 (*P*=0.0088 and 0.0003, respectively) (Figure 26). These microinvertebrate densities at Lock 1 were high for October in comparison to all other years (Figure 26). Microinvertebrate density at Lock 1 gradually increased with increasing temperature throughout the study period (Figure 26; Figure 28). Species richness was also significantly greater at Lock 1 in comparison to Lock 6 from the 6 October until the 2 November 2015 (*P*=0.0005–0.0163) (Figure 27). Species richness on the 2 November 2015 at Lock 1 was the highest for the sampling period (26±1.1 taxa) and was high in comparison to all other years excluding 2016-17 (Figure 27). As for the previous year, most microinvertebrates originated from the southern basin (Murray and tributaries) and were common species (Table 17). *Keratella javana* was likely from a northern tributary. Notable was the occurrence of *Keratella americana*, first record for the continent of this tropical species first described from the Americas.

2016-17

In 2016-17, a high flow year, microinvertebrate density (2,408 ±100) and species richness (34 ±1.18) was the highest for all four years (Figure 25; Figure 26; Figure 27). PERMANOVAs indicated that there was a significant interaction between site and trip for both density (*P*=0.0001) and species richness (*P*=0.0001), which suggests that differences in microinvertebrate composition among trips were not consistent between sites and vice versa.

From September until mid-December 2016, very small volumes of environmental water were released in comparison to unregulated flows (Figure 5). Increases in microinvertebrate density and species richness coincided with increases in discharge and water temperature at both Lock 6 and Lock 1 (Figure 25; Figure 26; Figure 27). This excluded the 6 December 2016 when microinvertebrate density and species richness was significantly lower than the preceding trip at both Lock 6 and Lock 1 (*P*=0.0001 and 0.0003, respectively for density, and *P*=0.0096 and 0.0003, respectively for species richness). At this time, dissolved oxygen levels had fallen below 2 mg/L at both sites due to the hypoxic effects of blackwater (Ye *et al.* 2018) which was the likely cause for reduction in density.

In late-December 2016, Commonwealth environmental water delivery in the LMR reduced the steepness of the flow recession and peaked on the 22 December 2016 at 8,100 ML/d (Figure 5). This coincided with the recovery of microinvertebrate density and species richness at both Lock 6 and Lock 1 on the 21 December 2016 to values similar to those observed prior to the blackwater event (Figure 25; Figure 27). Dominant taxa during December/January were a mix of Murray and Darling River taxa, primarily rotifers, including the tropical warm water brachionids *B. caudatus personatus* and *B. durgae*, the latter new to the continent (Table 17). The introduced *Keratella americana* was again recorded at both Lock 6 and 1. Also recorded during December/January were *Hexarthra braziliensis* a South American rotifer, and *Daphnia galeata*, a Holarctic species (Karabanov *et al.* 2018), both new to the continent.

Microinvertebrate density and species richness at Lock 6 and Lock 1 gradually increased with increasing temperature and discharge throughout the study period (excluding the 2 December 2016 during the blackwater event) (Figure 25; Figure 26; Figure 27).

2017-18

In 2017-18, microinvertebrate density varied between 97 (±17) and 1,552 (±57) ind/L, while species richness varied between 4.3 (±0.51) and 25 (±1.12) taxa (Figure 25; Figure 26; Figure 27). PERMANOVAs indicated that there was a significant interaction between trip and site for both density (P=0.0001) and species richness (P=0.0005), which suggests that differences among trips were not consistent between sites and vice versa.

From July to October 2017, environmental water (almost entirely Commonwealth environmental water) contributed to increases in discharge in the LMR from 4,500 to 8,700 ML/d in early September 2017 (Figure 5). This watering action coincided with the concurrent raising of Weir Pools 2 and 5 (August to October) (Appendix B). These combined actions did not correspond with high microinvertebrate density or species richness at either Lock 6 or Lock 1 at first sampling on 3 October 2017 (Figure 25; Figure 26; Figure 27). Predominant microinvertebrate taxa during this period were primarily southern Basin assemblages (Table 17)

From November 2017 to January 2018, environmental water (62% The Living Murray and 30% Commonwealth environmental water)) was delivered to the LMR, increasing discharge from 6,700 to 17,800 ML/d in early December 2017 (Figure 5). This watering action followed the raising of Weir Pools 7 (September to December), 8 (September to November) and 9 (September to October). At Lock 6, microinvertebrate density was significantly greater on the 30 October 2017 and the 13 November 2017 than the previous two sampling trips on the 3 October 2017 and 16 October 2017 (P=0.001–0.0018) and significantly greater than at Lock 1 (P=0.0005 and 0.015, respectively) (Figure 25; Figure 26). These higher microinvertebrate densities were largely due to high representation of littoral (facultatively pelagic) rotifers within the microinvertebrate community (Figure 25; Figure 27). Microinvertebrate density at Lock 1 gradually increased with increasing temperature throughout the study period (Figure 26). Species richness at both sites increased with increasing temperature and discharge (Figure 27). During this period, the microinvertebrate assemblage was a mixed, species-rich assemblage primarily of warm-water taxa from, for example, the Murrumbidgee and Lake Victoria, with cool-water species from the Goulburn/southern Basin.

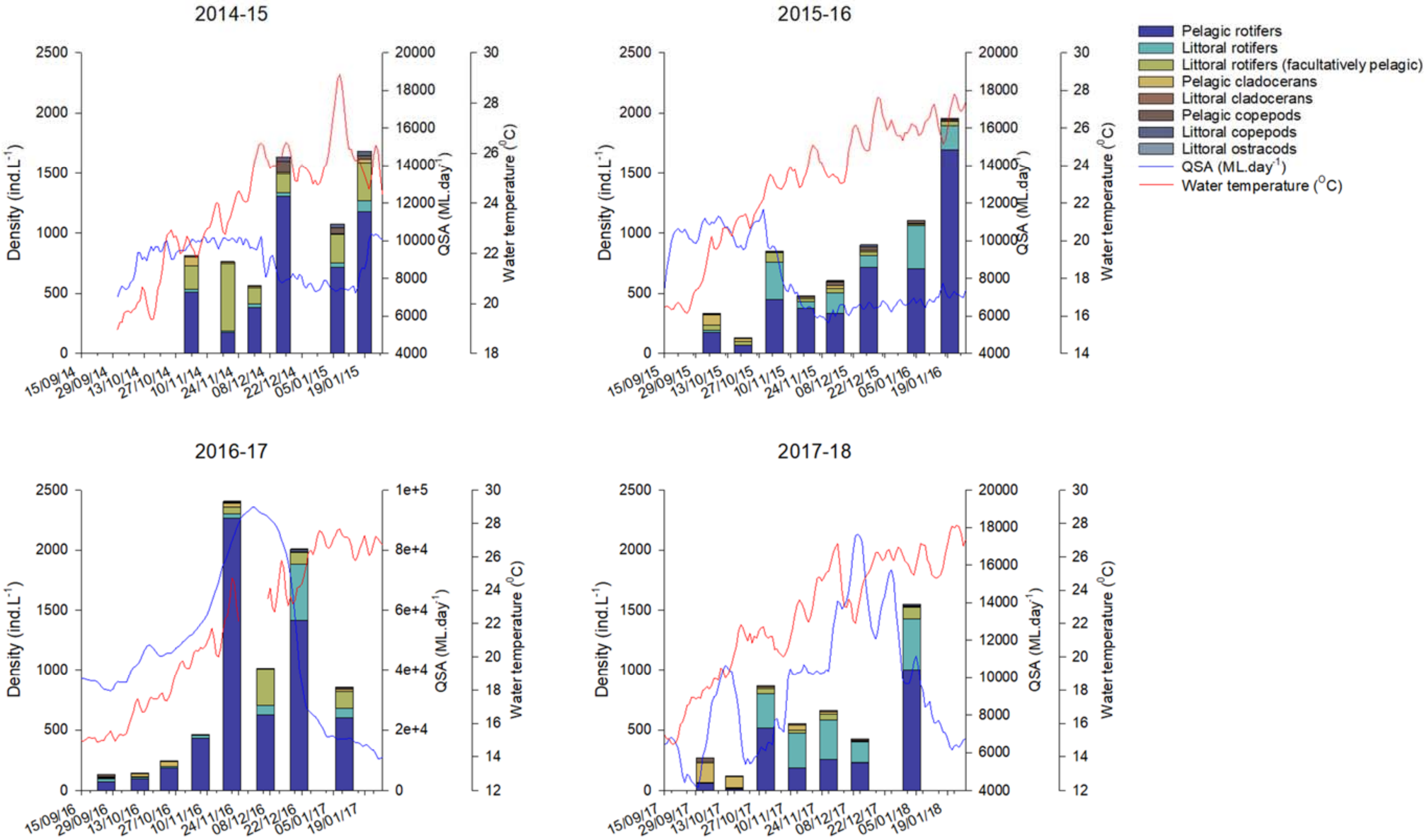

Figure 25. Mean microinvertebrate density collected in the LMR at below Lock 6 in the floodplain geomorphic zone, demonstrating community assemblage characteristics in a) 2014-15, (b) 2015-16, (c) 2016-17 and (d) 2017-18, plotted against flow discharge (blue line) and temperature at SA Border (red line).

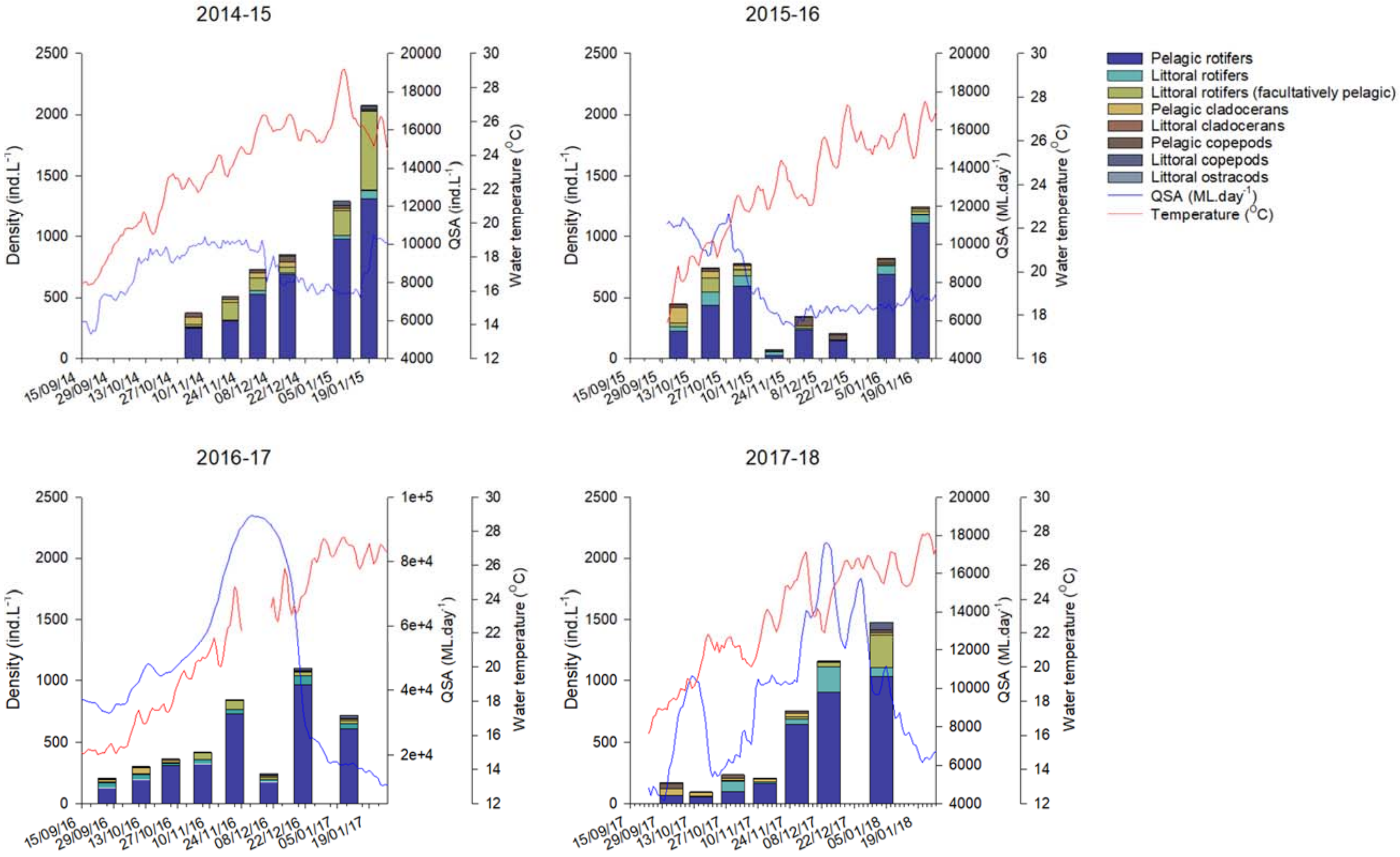

Figure 26. Mean microinvertebrate density collected in the LMR at below Lock 1 in the floodplain geomorphic zone, demonstrating community assemblage characteristics in a) 2014-15, (b) 2015-16, (c) 2016-17 and (d) 2017-18, plotted against flow discharge (blue line) and temperature at SA Border (red line).

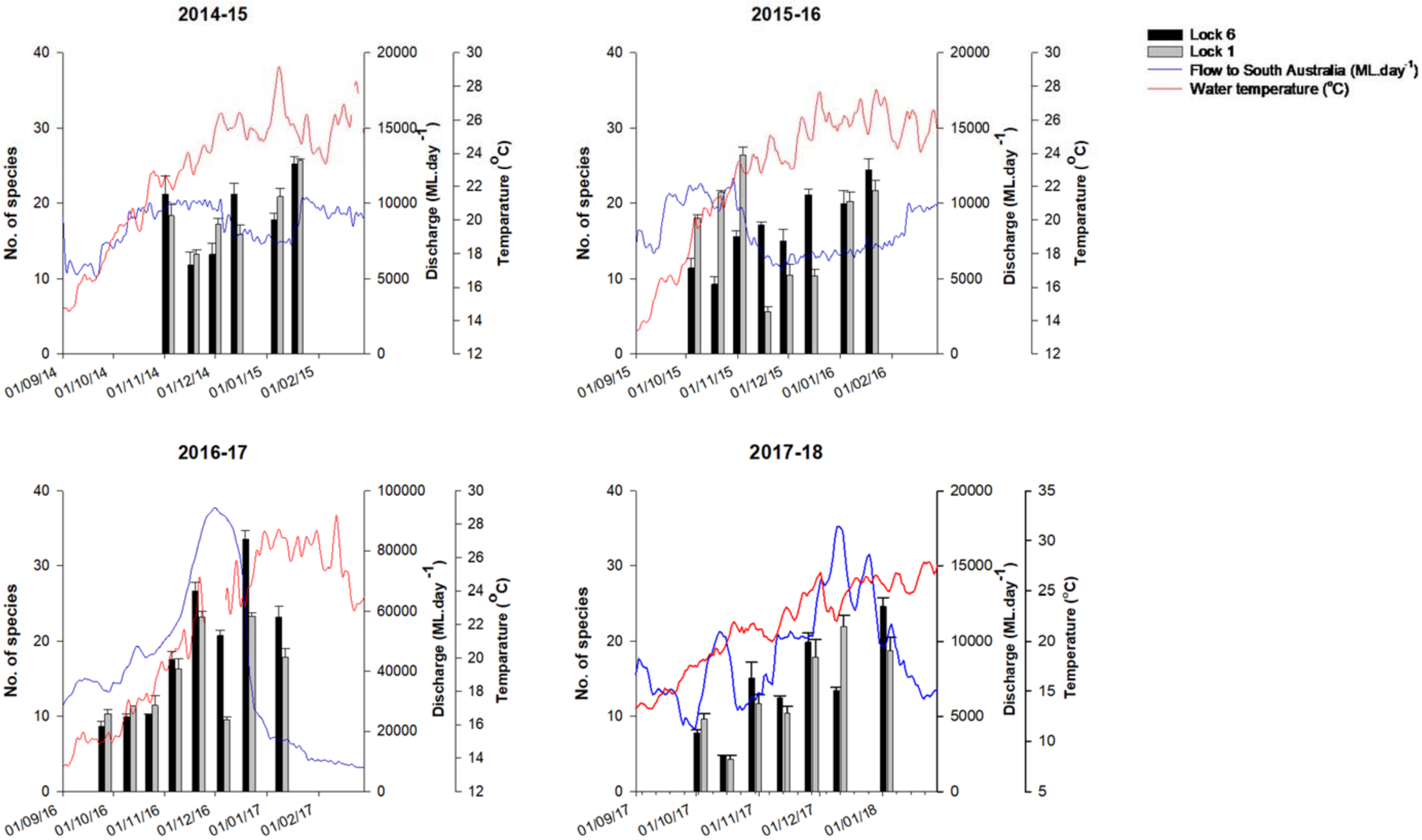

Figure 27. Mean (±S.E.) species richness of microinvertebrates collected in the LMR at sites below Lock 6 (black bars) and Lock 1 (grey bars).

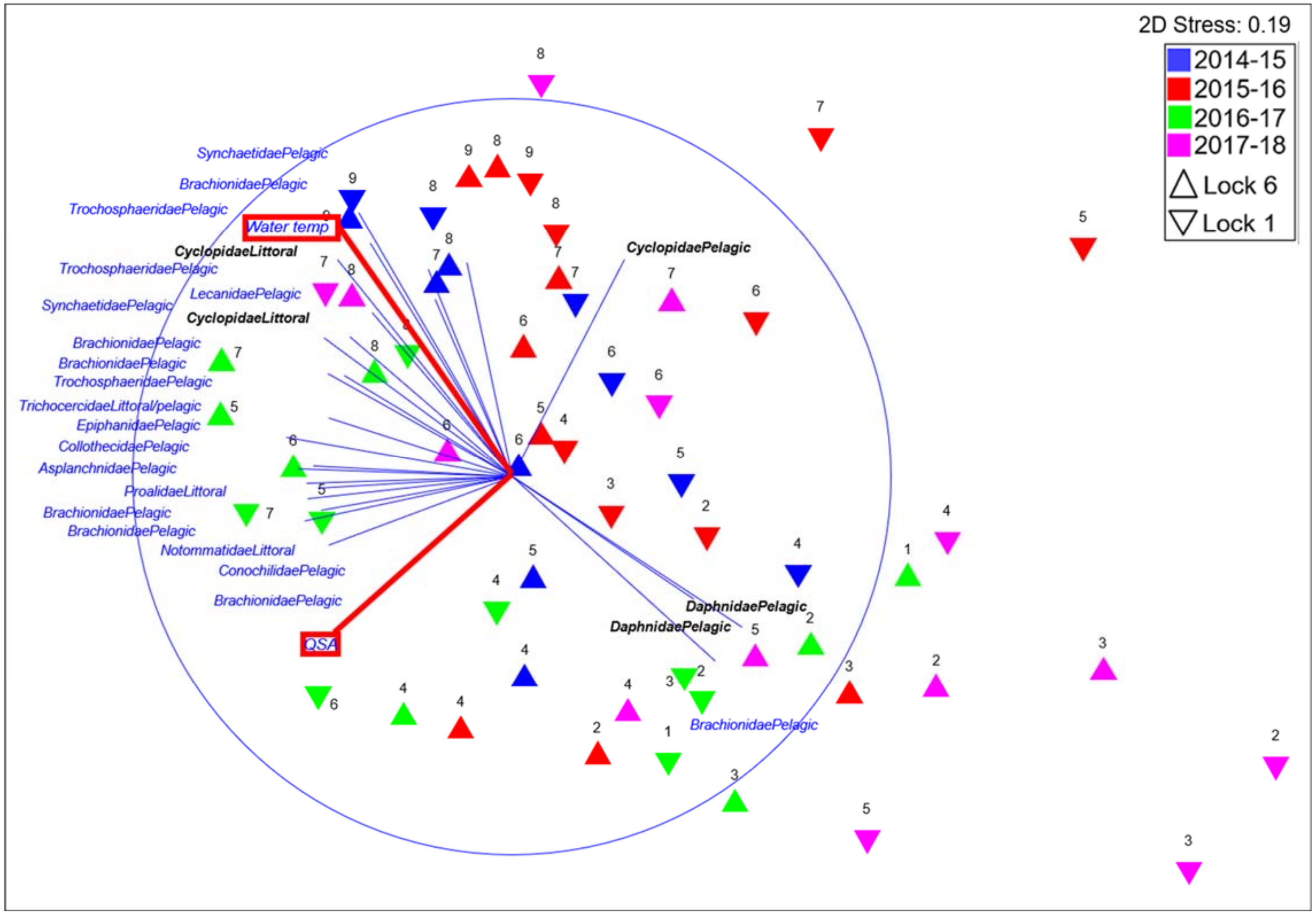

Figure 28. MDS ordination of microinvertebrate assemblage data (square-root transformed) at Lock 6 and Lock 1 during 2014–2018. Samples are labelled by sampling trip. Species density correlations are overlaid with blue vectors and water temperature (Water temp) and discharge into South Australia (QSA) correlations are overlain with red vectors. Species have been labelled by their family and preferred habitat (FamilyHabitat). Rotifer labels are blue and microcrustacean (cladocerans and copepods) labels are in black. Correlation value set to 0.5.

Table 17. Table summarising the timing and origin of environmental water delivered to the LMR in 2014-15, 2015-16 and 2017-18 and the species indicated by SIMPER analysis driving major differences in the microinvertebrate community due to higher densities at the time of delivery and likey community origin. The sampling period 2016-17 is not included due to the very low propotion of environmental water delivered during that period.

Year	**Timing**	**E-water origin**	**Species driving differences (due to higher density) in the microinvertebrate community in comparison to sampling trips that fall outside of the delivery period defined in column two.**	**Indicators of community origin**
2014-15	November (sampling trips 4 & 5)	Primarily from upstream of Yarrawonga, considerable contributions also from Lake Victoria and Victorian tributaries and a small percentage from the Murrumbidgee	*Filinia pejleri*, *Trichocerca similis grandis*, *Trichocerca pusilla*, *Synchaeta oblonga*, *Synchaeta pectinata*, *Polyarthra dolichoptera*, *Bosmina meridionalis*, *Trichocerca similis*, *Conochilus natans*, *Conochilus dossuarius* and *Filinia longiseta.*	All long-established taxa known from the Upper Murray and/or Goulburn systems (Shiel 1978; 1981)
	Late January (sampling trip 9)	Primarily from upstream of Yarrawonga and Lake Victoria, and a small percentage from Victorian tributaries the Murrumbidgee.	*Polyarthra dolichoptera*, *Keratella tropica*, *Filinia terminalis*, *Trichocerca pusilla*, *Anuraeopsis fissa*, *Trichocerca similis grandis*, *Brachionus diversicornis*, *Keratella lenzi*, *Hexarthra* sp., *Conochilus natans*, *Collotheca* cf. *tenuilobata*. *Filinia longiseta*, *Keratella tropica*, and *Brachionus falcatus.*	As above. e.g. *Filinia terminalis* is a cool water taxon and was likely derived from an Upper Murray and/or southern (Victoria) source.
2015-16	September until November (sampling trips 2, 3 &4)	Primarily from the Murray, considerable contributions also from the Murrumbidgee, small contributions from Victorian tributaries and very small contributions from the Lower Darling.	*Bosmina meridionalis*, *Filinia pejleri*, *Keratella javana*, *Synchaeta oblonga*, *Conochilus dossuarius*, *Trichocerca pusilla*, *Trichocerca similis*, *Polyarthra vulgaris*, *Conochilus natans*, *Keratella australis*, *Filinia grandis*. *Brachionus bidens*, *Keratella tropica*, *Trichocerca pusilla*, *Synchaeta oblonga*, *Synchaeta pectinata*, *Brachionus calyciflorus amphiceros* and *Cephalodella catellina.*	With the exception of *Keratella javana*, all are known from the Upper Murray catchment. *K. javana* was not collected in the cited earlier studies, is a known tropical species, and likely derived from a northern tributary,

Year	Timing	E-water origin	Species driving differences (due to higher density) in the microinvertebrate community in comparison to sampling trips that fall outside of the delivery period defined in column two.	Indicators of community origin
2017-18	July to October (sampling trips 2 & 3)	Return flows from the Goulburn and the Murrumbidgee rivers	*Bosmina meridionalis*, *Daphnia lumholtzi*, *Boeckella triarticulata*, *Ceriodaphnia* sp.	All common in the southern basin.
	November until January (sampling trips 4, 5, 6 and 7)	Return flows from Barmah-Millewa Forest and Hattah Lakes and a pulse from the Goulburn.	*Trichocerca agnatha*, *Synchaeta oblonga*, *Synchaeta pectinata*, *Conochilus natans*, *Conochilus dossuarius*, *Keratella tropica*, *Keratella americana*, *Filinia terminalis*, *Trichocera similis grandis*, *Polyarthra dolichoptera*, *Bosmina meridionalis*, *Hexarthra intermedia*, *Collotheca pelagica* and *Polyarthra vulgaris.*	Again, well-established southern basin taxa, with the exception of the introduced *Keratella americana*, native to the Americas, and not recorded in Australia prior to LTIM. Source of introduction unknown. Possibly as a hitch-hiker with stocked fish or via the pet/aquarium trade, or (less-likely) bird-vectored propagules.

Larval gut content

From 2014–2018, there was a larger proportion of Murray cod larvae with empty guts (69–100%), compared to freshwater catfish (20–57%) and golden perch (0–23%) (Table 13). The larval diet of the three species was comprised mostly of copepods and cladocerans (Figure 29; Figure 30). The pelagic calanoid copepod *Boeckella triarticulata* was the most frequent prey item for all species and was numerically abundant in the diet of all species, particularly golden perch. Whilst only present in guts during one year, the cladocerans *Ceriodaphnia cornuta* (pelagic) and *Picripleuroxus quasidenticulatus* (littoral) were abundant prey for Murray cod and freshwater catfish, respectively. The pelagic cladoceran *Bosmina meridionalis* and benthic atyid shrimp were numerically dominant in the diet of golden perch and freshwater catfish, respectively.

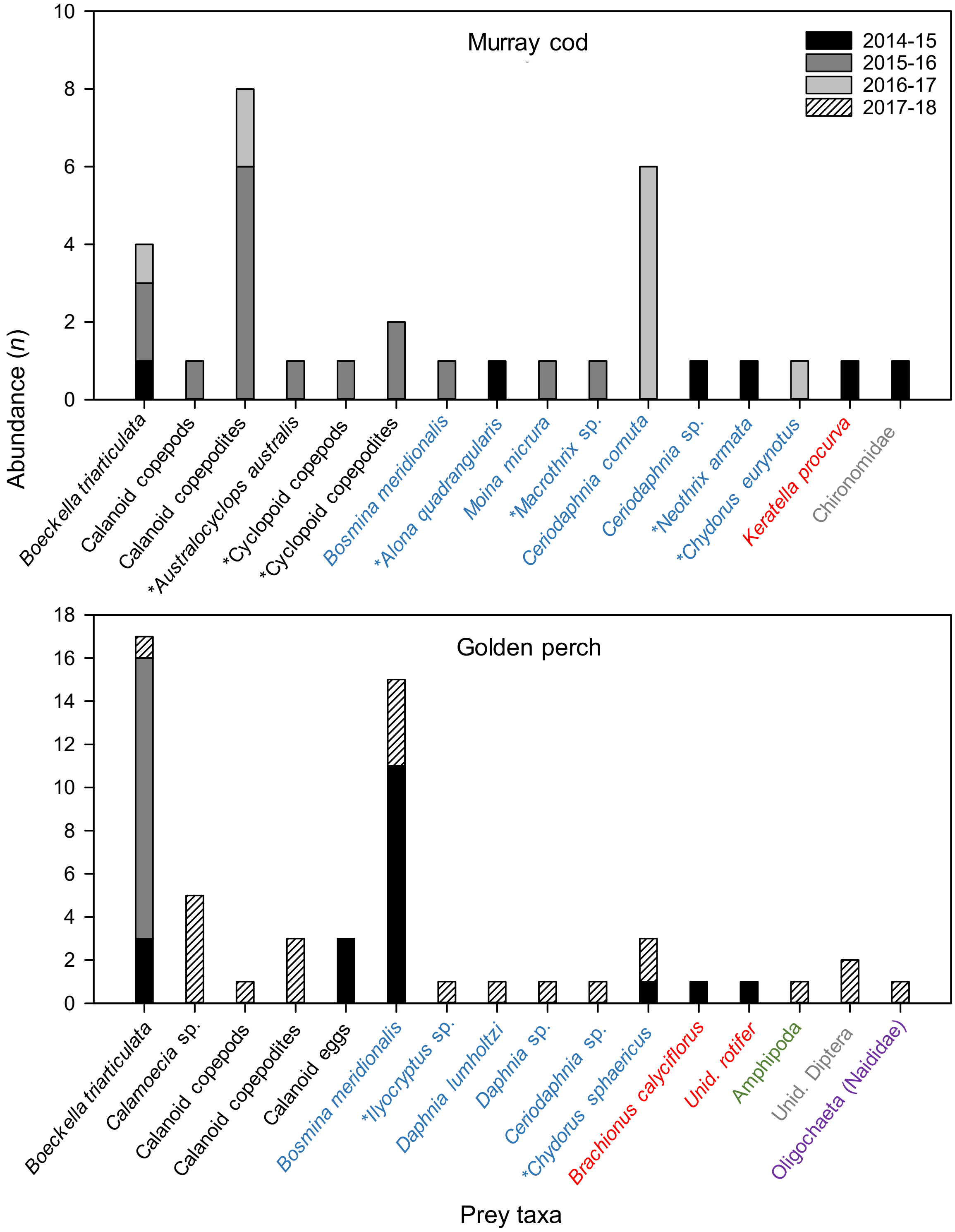

Figure 29. Abundance (number, *n*) of prey identified in the guts of post-flexion Murray cod (*n* = 15; TL = 7.8–12.0 mm) and golden perch (*n* = 13; TL = 6.5–20 mm) from 2014–2018. Prey taxa are presented as copepods (black), cladocerans (blue), rotifers (red), malacostracans (green), insects (grey) and oligochaetes (purple). * indicates littoral microcrustaceans.

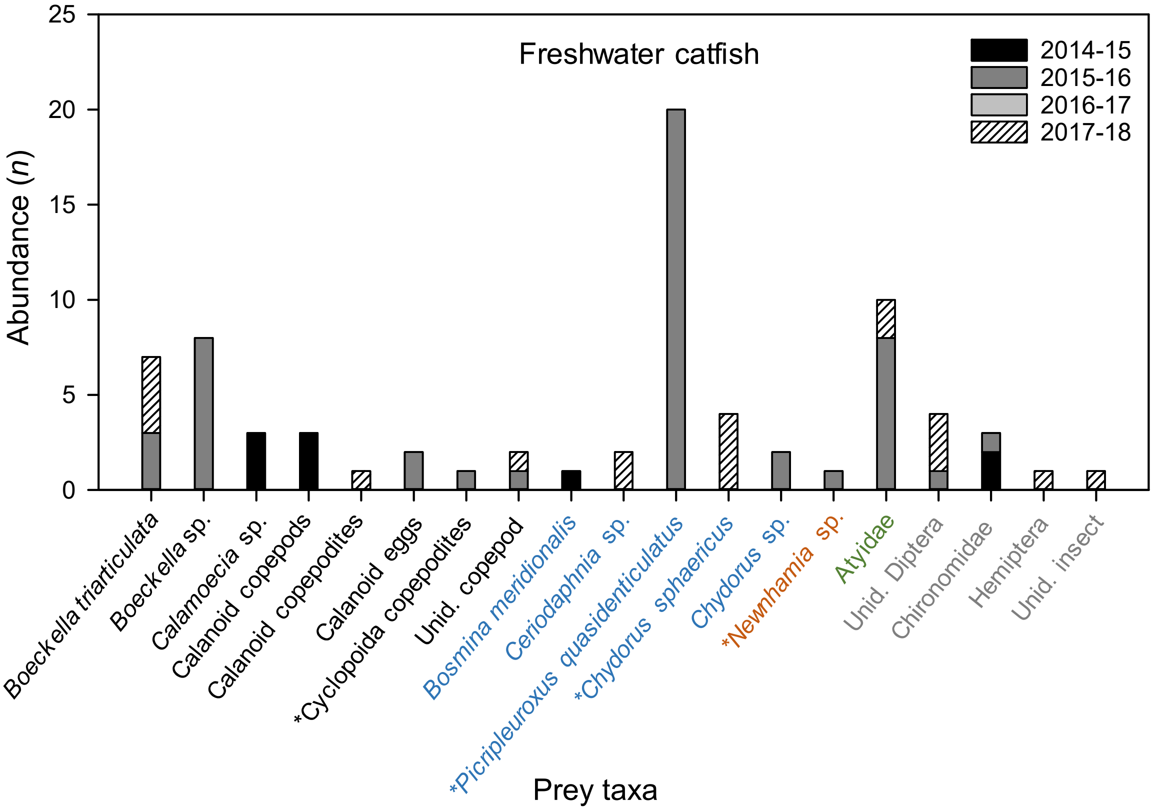

Figure 30. Abundance (number, *n*) of prey identified in the guts of post-flexion freshwater catfish (*n* = 20; TL = 13.0–20.7 mm) from 2014–2018. Prey taxa are presented as copepods (black), cladocerans (blue), ostracods (orange), malacostracans (green) and insects (grey). * indicates littoral microcrustaceans.

Evaluation

The evaluation approach, including assessment criteria, is described in the evaluation section for hydrological regime (Section 2.1).

Table 18. Microinvertebrates evaluation questions and answers. CEW = Commonwealth environmental water, eWater = environmental water.

CEWO evaluation questions	**Outcomes of CEW delivery**			
	2014-15	**2015-16**	**2016-17**	**2017-18**
What did CEW contribute to microinvertebrate diversity?	25.8 ±0.2 taxa*	26±1.1 taxa*	34±1.2 taxa.	25±1.1 taxa*
	Commonwealth environmental water contributed to increases in discharge that coincided with increases in intra-annual species richness in the LMR in most years. The number of distinct eWater releases were considered and the proportion of those events in which increases in diversity were achieved at one or both sites. E.g. in 2014-15, there were two distinct eWater releases. Both releases resulted in increases in diversity at both Lock 6 and Lock 1. Maximum mean species richness ± standard error is provided for each sampling year as a response to the overall flow regime of that year (not necessarily associated with CEW).			
What did CEW contribute via upstream connectivity to microinvertebrate communities of the LMR?	29%	42%	32%	44%
	CEW contributed to longitudinal connectivity and most likely the transport of still- (heleoplanktonic), warm-water taxa, including novel taxa for the LMR or the continent, to the LMR in January 2017 and January 2018. These could have derived from northern tributaries, or from populations established in Lake Victoria. For most years of LTIM, longitudinal connectivity of river flow was indicated by the transportation of microinvertebrate taxa, known only from upstream catchments (i.e. Murray and its tributaries), to the LMR (e.g. cool water taxa such as *Filinia terminalis* likely originated from Goulburn sources in 2015-16). With the exception of 2016-17, when high flows were unregulated and eWater was not delivered, the transport of mixed assemblages from different upstream Murray River sources coincided with upstream CEW watering events. It is difficult, however, to clearly distinguish between CEW and non-CEW delivery sources. As planktonic (pelagic) taxa must come from upstream in a flowing system, the proportion of planktonic taxa are provided for each sampling year as a response to the overall flow regime of that year (not necessarily associated with CEW).			

CEWO evaluation questions	**Outcomes of CEW delivery**			
	2014-15	**2015-16**	**2016-17**	**2017-18**
What did CEW contribute to microinvertebrate density?	2,076±117 ind/L*	1,955±205 ind/L*	2,408±100 ind/L	1,552±57 ind/L*
	CEW, in combination with additional management levers, coincided with increases in intra-annual microinvertebrate density in most years. The number of distinct eWater releases were considered and the proportion of those events in which increases in density were achieved at one or both sites. Maximum mean density ± standard error is provided for each sampling year as a response to the overall flow regime of that year (not necessarily associated with CEW).			
What did CEW contribute to the timing and presence of key species in relation to the diet of large-bodied native fish larvae?#	It is difficult to tie the presence of large-bodied fish larvae prey with specific flow deliveries or sources, particularly since some prey species, which were abundant in the guts of larvae during a certain year, were not collected during ambient sampling in that same year (e.g. *Calamoecia* sp. in golden perch 2017-18). The low number of fish larvae sampled also limits the reliability of results. Therefore, the contribution of CEW on the dietary composition of large-bodied fish larvae could not be evaluated.			

* = additional management levers were also used (Appendix B)

= The collection of larval fish (Section 2.5) provided an opportunity to explore the link between eWater and fish diet (microinvertebrates). This was an additional evaluation question that was added to the indicator in a cost-effective manner.

Contribution (to what extent CEW contributed towards the outcome, with the significance of the outcome considered):
⬜ Unknown 🟥 Negative ⬜ None/negligible 🟩 Minor 🟩 Moderate 🟦 Substantial

Discussion

The microinvertebrate assemblage demonstrated consistent seasonal trends across years, with variability between years associated with variability in discharge. During low flow years, seasonal changes were primarily driven by changes in the dominance of pelagic rotifers. This was evident in 2014-15 and 2015-16. During periods of higher discharge, these changes still occurred, however more littoral and pelagic organisms played a role in driving community assemblages. This was particularly evident in 2016-17, but also in 2017-18 during a higher in-channel flow pulse than those in 2014–2016 in the LMR. Therefore, introducing variability in discharge increased variability in inter-annual microinvertebrate assemblage structure. There are a number of reasons as to why this variability may be important for the aquatic food web. Firstly, these increases in discharge increased the presence of littoral and pelagic organisms. Increases in these organisms during higher discharge indicates greater availability of food resources for higher trophic organisms. It also indicates enhanced lateral and longitudinal connectivity and thus dispersal, a crucial process in the maintenance of species and genetic diversity. The combination of dispersal and diversity also improves successional processes following disturbance (resilience) through the supply of new recruits both spatially (by transferring organisms into downstream habitats) and temporally (by increasing egg bank diversity). Finally, some studies have also found correlations between lower trophic level diversity (temporal and spatial) and enhanced productivity and community stability (e.g. Striebel *et al.* 2012) as well as enhanced diversity in higher trophic organisms (e.g. Eadie and Keast 1984). Interestingly, in late September and early October 2017-18, there were high densities of pelagic cladocerans, important food resources for Murray cod, golden perch and silver perch (Barlow *et al.* 1987; Arumugam and Geddes 1996; Warburton *et al.* 1998; Ingram and De Silva 2007; Kaminskas and Humphries 2009). It is possible that these higher cladoceran densities were due to a combination of egg bank replenishment, achieved during high discharge in 2016-17, and a small spring pulse (up to ~10,000 ML/d) that was enhanced by Commonwealth environmental water. Nevertheless, the early timing of these food resource increases were unlikely to benefit golden perch (and Murray cod) larvae, which had later spawn dates (Figure 34, Section 2.5).

The collection of larval fish (Section 2.5) provided an opportunity to explore the link between environmental water and fish diet (microinvertebrates). Unfortunately, the contribution of Commonwealth environmental water to the diet composition of large-bodied fish larvae could not be evaluated. Nevertheless, findings from this study add to our understanding of the larval diets of these native species. Whilst a mixture of pelagic and littoral prey species were consumed by all species, the diets of Murray cod and golden perch larvae were dominated by common pelagic calanoid copepods (e.g. *Boeckella triarticulata*) and cladocerans (e.g. *Bosmina meridionalis* and *Ceriodaphnia cornuta*). Freshwater catfish consumed larger quantities of littoral microcrustaceans (e.g. *Picripleuroxus quasidenticulatus*) and benthic decapods (i.e. atyid shrimp). In contrast to this study, littoral microcrustaceans were important in the diet of Murray cod larvae in the Chowilla anabranch (Gibbs *et al.* 2020), which has narrower channels and increased lateral connection. This suggests that environmental water delivery that increases lateral connectivity will enhance food resources for large-bodied fish larvae by directly increasing the abundance of littoral microcrustacean prey in the main channel.

Commonwealth environmental water contributed to increases in discharge that coincided with increases in intra-annual species richness in the LMR in most years. This included in October (in conjunction with the operation of the Chowilla regulator) and January 2014-15, in October and November 2015-16 (in conjunction with weir pool raising), and throughout the majority of the sampling period in 2017-18. Connectivity via Chowilla floodplain in 2014 and weir pool manipulation in 2015 likely promoted hatching events, leading to observed population density and/or species richness increases, in part due to increases in littoral organisms. The relationship between discharge and species richness was also evident in 2016-17 when the highest species richness for the entire study period occurred during peak flood. Different microinvertebrate species and/or groups of species are likely to support different higher trophic organisms and/or life history stages. This is because morphological and behavioural characteristics of prey can be restrictive to predators, where certain prey types may be restrictive to one predator, however not another (Vinyard and O'Brien 1975; Cooper and Goldman 1980; Dodson and Egger 1980; Ranta and Nuutinen 1985; Mills *et al.* 1986; Schael *et al.* 1991; Bremigan and Stein 1994). For example, at times increases in species richness were in part driven by increases in littoral organisms, e.g. in October and November at Lock 6 in 2014-15 and in October and early-November 2015 at Lock 1. Species that prefer pelagic environments are often translucent (e.g. *Synchaeta* species) and/or are good swimmers in the open water. In comparison, littoral organisms are often darker in colour and/or poorer swimmers, limiting their ability to avoid predation when in the open water (e.g. species from the family *Chydoridae*) (e.g. Lair 2006). This may be particularly important for larval/juvenile fish with poor eyesight and/or poor swimming ability. Therefore, these increases in richness of prey options during increased discharge (including the delivery of environmental water) are likely to have increased the range of higher trophic organisms and life history stages which the microinvertebrate community can support.

A significant proportion (>90%) of recorded microinvertebrate taxa in the plankton of the LMR has been established in Murray River tributaries of the southern basin over a very long time frame, given the long evolutionary history of the MDB. It is from these established populations that increased river flow, including Commonwealth environmental water, effectively 'restocks' depleted microinvertebrate populations in the LMR, particularly after long periods of low or no flow. Peak microinvertebrate species richness below Lock 6 was observed during high flows in 2016-17 when different water sources, primarily the Murray and its tributaries, contributed to discharge to the LMR. Taxa known only from those catchments that were recorded in the LMR indicate increased longitudinal connectivity. A small proportion of taxa recorded in the LMR were warm stenothermal 'tropical' species not expected to persist in the cooler waters of the southern basin. These coincided with small late-summer Darling environmental flows, and likely originated from the Darling River headwaters.

Commonwealth environmental water, in combination with additional management levers, coincided with increases in intra-annual microinvertebrate density in some years. These management levers included the operation of the Chowilla regulator in 2014-15 and 2015-16 that coincided with increases in density at Lock 6, and the raising of Weir Pools 2 and 5 in 2015-16 and the raising of Weir Pools 7, 8 and 9 in 2017-18, which coincided with increases in density at Lock 1 and Lock 6, respectively. In 2016-17, environmental water was also delivered to mitigate the impacts of a severe blackwater event (Figure 5). Prior to this release, the low dissolved oxygen levels had resulted in a significant decline in the density of microinvertebrates at both Lock 6 and Lock 1. Multicellular rotifers and microcrustaceans experienced high mortality. The environmental water releases may have contributed to the recovery of microinvertebrate densities in late-December 2016. Increases in density have positive outcomes for the broader food web as microinvertebrates provide food for a range of higher trophic organisms and the greater the density of microinvertebrates, the higher the rate of predator-prey encounters. Therefore, predators aren't required to expend as much energy to consume adequate food for their own physiological and behavioural requirements (Cooper and Goldman 1980; Vinyard 1980). Not surprisingly, microinvertebrate density also demonstrated relatively consistent increases with increasing water temperature at one or both sites in all years. This increased productivity with temperature was most likely due to greater primary productivity and therefore greater food availability, in combination with higher growth rates (e.g. Betsill and Van Den Avyle 1997). Therefore, despite at times environmental water delivery not appearing to increase density beyond what was expected at specific times of the year, there was no evidence of dilution effects. Therefore, these environmental water deliveries were contributing to considerable increases in the overall quantity of microinvertebrates available to higher trophic organisms during the warmer months.

Management implications

Environmental water delivery, combined with river management such as weir pool manipulation and the operation of the Chowilla regulator, can potentially contribute to increases in microinvertebrate density and species richness, overall quantity of microinvertebrates available and facilitate long distance dispersal, as indicated on multiple occasions during the study period. Therefore, delivering environmental water at times known to be important for food-web productivity, e.g. spring, is likely to have a positive influence on a diverse range of higher trophic organisms, as increases in microinvertebrate species richness implies an increase in the diversity of available food resources to a range of higher trophic organisms and/or life history stages. Furthermore, environmental water delivery at times known to be important for zooplanktivorous fish, is likely to improve energy transferal to higher trophic organisms, as increases in microinvertebrate density is likely to increase the rate of predator-prey interactions. Lastly, environmental water delivery that contributes to increasing discharge in warmer months is likely to contribute to considerable increases in the overall quantity of microinvertebrates available to higher trophic levels due to the consistent positive relationship between increasing water temperature and microinvertebrate density and minimal evidence of dilution effects.

Similar management approaches however did not always have the same results at similar times of the year. This highlights the need for a greater understanding of what is driving the different outcomes so that consistent results can be achieved under similar flow scenarios. The primary flow scenario shaped by environmental water throughout the LTIM Project was the 'spring pulse', which naturally would have occurred on almost an annual basis (e.g. Figure 5). The 'spring pulse' is believed to be linked to key ecological processes in the LMR and is critical for the riverine productivity and life history strategies of many native fish species, e.g. spawning and recruitment. Thus, improving our understanding of flow effects on productivity and the ability to recreate increases in density and species richness of microinvertebrates (potential prey) consistently, through further monitoring and investigations, may have significant long-term positive outcomes for native fish and the aquatic food web in the LMR.

Conclusion

The delivery of environmental water, combined with the use of management levers such as weir pool manipulation and the operation of the Chowilla regulator, coincided, at times, with increases in species richness, density and dispersal during the LTIM Project. Although, due to concurrence of multiple events and factors, the mechanisms facilitating these increases are not fully understood. Increased species richness and density are likely to have had short-term (through the provision of an abundance and variety of food resources) and long-term (through the replenishment and diversification of the egg-bank) outcomes, which with further monitoring and improvement will support the rebuilding of resilience into the aquatic food web throughout the Murray River.

2.5 Fish Spawning and Recruitment

Background

Spawning and recruitment of golden perch in the southern MDB corresponds with increases in water temperature and flow, either in-channel or overbank (Mallen-Cooper and Stuart 2003; Zampatti and Leigh 2013a; 2013b). Flow and hydraulic variability is a cue for spawning, but it may also enhance recruitment through increased larval dispersion and riverine productivity. Silver perch display similar life history characteristics and population dynamics, although in the lotic (flowing) reaches of the Murray River, silver perch may spawn circa-annually (Tonkin *et al*. 2017). Due to these flow-related traits, golden perch and silver perch are candidates for measuring ecological response to environmental water allocations. Understanding the influence of hydrology on the population dynamics of golden perch and silver perch, however, is reliant on accurately determining the hydrological conditions at the time and place of crucial life history processes. For example, to be able to accurately associate river flow with spawning, the time and place of spawning must be known.

Natural flow regimes in the southern MDB have been substantially altered by river regulation and increased extraction, leading to decreased hydrological (e.g. flow) and hydraulic (e.g. water level and velocity) variability, and reduced floodplain inundation (Maheshwari *et al.* 1995; Bice *et al.* 2017). From 2014–2019, >500 GL/year of Commonwealth environmental water was delivered to the LMR to augment flow regimes in the southern MDB to rehabilitate the health of aquatic ecosystems. In the LMR, Commonwealth environmental water was primarily used to contribute to increased base flows and freshes (i.e. increases in flow contained within the river channel) (Section 1.4). Through the delivery of these flows, one of the ecological objectives was to contribute to increased spawning and/or recruitment of flow-dependent fish species.

Over the five-year term of this project, we aimed to identify potential associations between reproduction (spawning and recruitment) of golden perch and silver perch and environmental water delivery. Our specific objective was to identify timing of spawning and the source (i.e. natal origin) of successful recruits to enable association of ecological response with hydrology, and to explore population connectivity between regions of the southern connected MDB.

Hypotheses

- Increased flow (nominally >15,000 ML/d, Zampatti and Leigh 2013a) in spring–summer will promote the spawning and recruitment (to young-of-year, YOY) of golden and silver perch.
- Multiple years of enhanced spring–summer flow will increase the resilience of golden perch and silver perch populations in the LMR.

Methods

To evaluate the contribution of Commonwealth environmental water to the spawning and recruitment of golden perch and silver perch in the LMR from 2014–2019, we: (1) sampled larval and young-of-year (YOY) fish at sites in the gorge and floodplain geomorphic zones of the LMR (Figure 1); (2) used otolith microstructure and chemistry, specifically strontium (Sr) isotope ratios (^{87}Sr/^{86}Sr), to retrospectively determine the time and place of spawning; and (3) used electrofishing to collect a representative subsample of the golden perch and silver perch populations in the LMR to determine population age structure. Due to low numbers of silver perch collected during this project, we only present findings for golden perch in this report.

Analysis of water ^{87}Sr/^{86}Sr at sites across the southern MDB

To determine spatio-temporal variation in water strontium (Sr) isotope ratios (^{87}Sr/^{86}Sr) over spring–summer (early September to early February) of 2014–2019, water samples were collected weekly–monthly from approximately 11 sites across the southern MDB (Figure 31; Table 19). Water sample filtration and strontium isotope analysis procedures are described in full in Ye *et al.* (2016b).

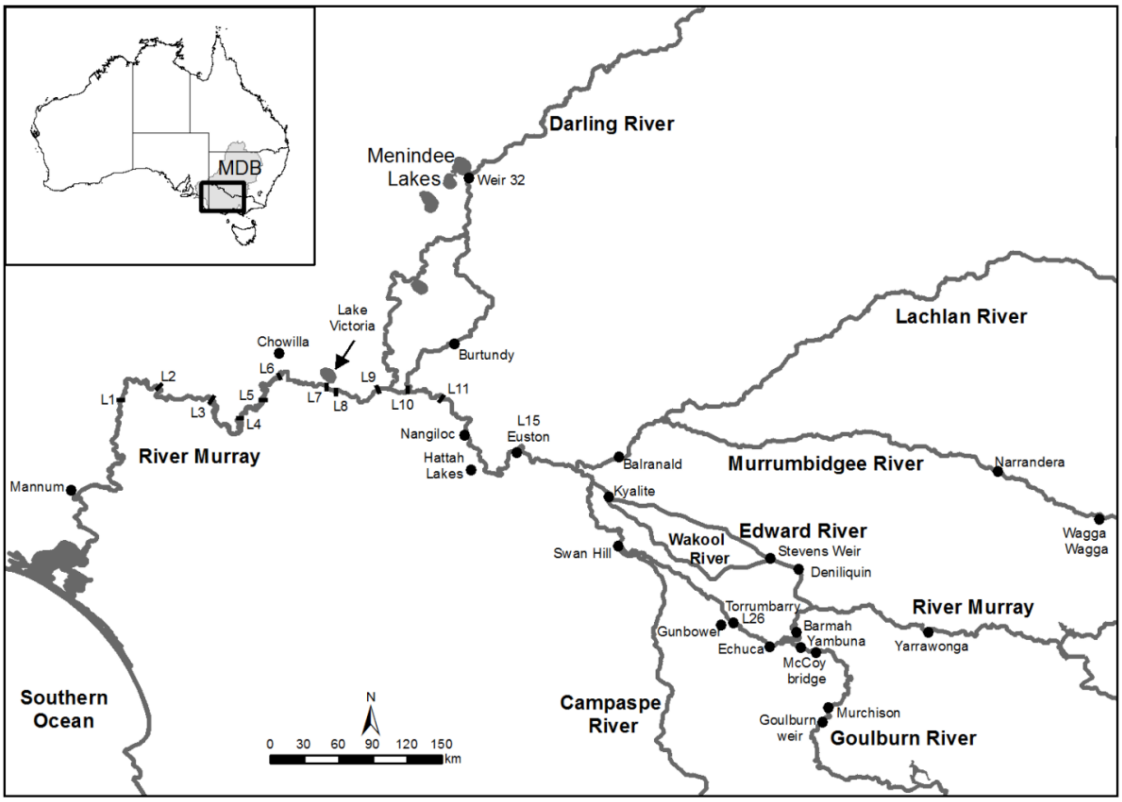

Figure 31. Map showing the location of the Murray–Darling Basin and the major rivers that comprise the southern Murray–Darling Basin, the numbered Locks (L) and Weirs (up to Lock 26, Torrumbarry), the Darling, Lachlan, Murrumbidgee, Edward–Wakool, Campaspe and Goulburn rivers and Lake Victoria, an off-stream storage used to regulate flows in the Murray River.

Table 19. Locations of water sample collection for ^{87}Sr/^{86}Sr analysis from 2014–2019.

River	Location
Murray	Lock 1, Lock 6, Lock 9, Lock 11, Torrumbarry, Barmah
Darling	Weir 32
Edward–Wakool	Deniliquin
Murrumbidgee	Narrandera
Goulburn	Yambuna, Pyke Road

Sampling eggs and larvae

From 2014–2019, larval fish sampling was conducted approximately fortnightly between early October and late January in the LMR (Table 20). Sites were located 5, 7 and 9 km downstream of Locks 1 and 6 (Figure 1). Three day-time and three night-time plankton tows were undertaken on the same day at sites 5 km below each lock, while one day-time plankton tow was undertaken at all other sites. Fish were preserved (70–95% ethanol) in the field and returned to the laboratory for processing. Golden perch and silver perch eggs are able to be visually differentiated from other fish eggs, but not from each other. When perch eggs were present, they were enumerated and a subsample were transported to the laboratory and hatched out to confirm the species.

Table 20. Larval fish sampling details from 2014–2019.

Year	Number of trips	Date range
2014-15	6	3/11/14 – 20/1/15
2015-16	8	6/10/15 – 21/1/16
2016-17	8	26/9/16 – 11/1/17
2017-18	7	3/10/17 – 4/1/18
2018-19	7	8/10/18 – 9/1/19

Sampling YOY and population age-structure

From 2015–2019, adult and juvenile golden perch were sampled using a 7.5 kW Smith Root (Model GPP 7.5) boat electrofishing unit at approximately 16 sites in the LMR (Ye *et al.* 2016b; 2017; 2018; 2019). Annual sampling was undertaken in April–May to complement Category 1 Fish Assemblage sampling and to maximise the likelihood of collecting YOY from the spring–summer spawning season. In 2017, however, electrofishing was delayed to winter due to equipment failure/malfunction.

Electrofishing was conducted during daylight hours and all available littoral habitats were surveyed. At each site, the total time during which electrical current was applied ranged from approximately 676 to 2880 seconds. All individuals were measured to the nearest mm (total length, TL) and a subsample of golden perch proportionally representing the length-frequency of golden perch collected was retained for ageing.

Ageing

Larvae/YOY golden perch were measured to the nearest millimetre and sagittal otoliths were removed. To estimate the spawn date of larval and YOY golden perch, daily increments in otolith microstructure were examined. Otoliths were mounted individually in Crystalbond™, proximal surface downwards, and polished down to the primordium using a graded series of wetted lapping films (9, 5, and 3 µm). Sections were then polished using 0.3 µm alumina slurry to a thickness of 50–100 µm. Sections were examined using a compound microscope (x 600) fitted with a digital camera and Optimas image analysis software (version 6.5, Media Cybernetics, Maryland, USA). Estimates of age were determined by counting the number of increments from the primordium to the otolith edge (see Ye *et al.* 2016b).

We investigated length and age-frequency distributions to assess the age structure and year-class strength of golden perch. Golden perch retained for ageing were euthanised and sagittal otoliths were removed. Whole otoliths were embedded in clear casting resin and a single 400 to 600 µm transverse section was prepared. Sections were examined using a dissecting microscope (x 25) under transmitted light. Estimates of age were determined independently by three readers by counting the number of discernible opaque zones (annuli) from the primordium to the otolith edge. YOY (<1 year old) fish were defined as individuals lacking clearly discernible annuli.

Otolith $^{87}Sr/^{86}Sr$ analysis, natal origin and migration history of golden perch

Sagittal otoliths were dissected and mounted individually in Crystalbond™, proximal surface downwards, on an acid-washed glass slide and polished down to the primordium using a graded series of wetted lapping films (9, 5 and, 3 µm). The slide was then reheated and the polished otolith transferred to a 'master' slide, on which otoliths from all collection sites were combined and arranged randomly to remove any systematic bias during analysis. The samples were rinsed in Milli-Q water (Millipore) and air dried overnight in a class 100 laminar flow cabinet at room temperature. *In situ* microsampling analysis of $^{87}Sr/^{86}Sr$ in the otoliths of larval, juvenile and adult golden perch was achieved by laser ablation – inductively coupled plasma mass spectrometry (LA-MC-ICPMS). The laser ablation and mass spectrometry procedure is described in full in Ye *et al.* (2016b).

To investigate the natal origin and migration history of new recruits (larvae and YOY) and dominant cohorts of golden perch in the LMR from 2015–2019, we analysed $^{87}Sr/^{86}Sr$ from the otolith core to edge in a subsample of fish. We compared these transects to water $^{87}Sr/^{86}Sr$ measured at sites across the southern MDB from 2011–2019 (Zampatti *et al.* 2015; this report; SARDI unpublished data).

Data analyses

To compare golden perch egg and larval abundances across years, including years prior LTIM, we integrated data from other projects (e.g. Commonwealth environmental water Short-Term Intervention Monitoring, Ye *et al.* 2015a; 2015b; 2016a) and calculated catch-per-unit-effort (CPUE, individuals per trip) ± standard error. CPUE was calculated using two long-term sites: 5 km downstream Lock 1 and 5 km downstream Lock 6, between early October and late January. Data were represented graphically as bar charts.

Temporal variability in the relative abundance of golden perch was investigated by assessing changes in Category 1 electrofishing CPUE data (See Section 2.6, Fish Assemblage). Differences in the relative abundance (individuals per 90 seconds of electrofishing on-time effort) sampled between years at each site were analysed using univariate single-factor PERMANOVA (permutational ANOVA and MANOVA) in the software package PRIMER v. 6.1.12 and PERMANOVA+ (Anderson *et al.* 2008). These analyses were performed on untransformed, standardised relative abundance data.

Results

Water $^{87}Sr/^{86}Sr$ and hydrology

From 2011–2019, water $^{87}Sr/^{86}Sr$ was measured at sites across the southern MDB. Throughout this period, water $^{87}Sr/^{86}Sr$ remained reasonably stable in the Darling River and the Murray River and its tributaries upstream of the Darling River junction. Water $^{87}Sr/^{86}Sr$ was highest in the Murray River at Barmah and Torrumbarry (>0.7180), and lowest in the Darling River (<0.7080) (Figure 32a). Water $^{87}Sr/^{86}Sr$ generally decreased longitudinally along the Murray River, particularly during low Darling River flow contribution, as tributaries with distinct and relatively temporally stable $^{87}Sr/^{86}Sr$ (e.g. Goulburn River) contribute to flow (Figure 6).

Water $^{87}Sr/^{86}Sr$ in the lower River Murray (Lock 9–Lock 1) was temporally variable, with water $^{87}Sr/^{86}Sr$ generally decreasing and becoming more similar with greater spatio-temporal contributions of Darling River flow (e.g. 2011-12 and 2012-13) (Figure 32b). In contrast, during years of negligible Darling River flow (i.e. 2014-15, 2015-16 and 2018-19), variation in water $^{87}Sr/^{86}Sr$ across sites in the LMR increased, while water $^{87}Sr/^{86}Sr$ at Lock 9 was similar to water $^{87}Sr/^{86}Sr$ at Lock 11 in the mid-Murray River.

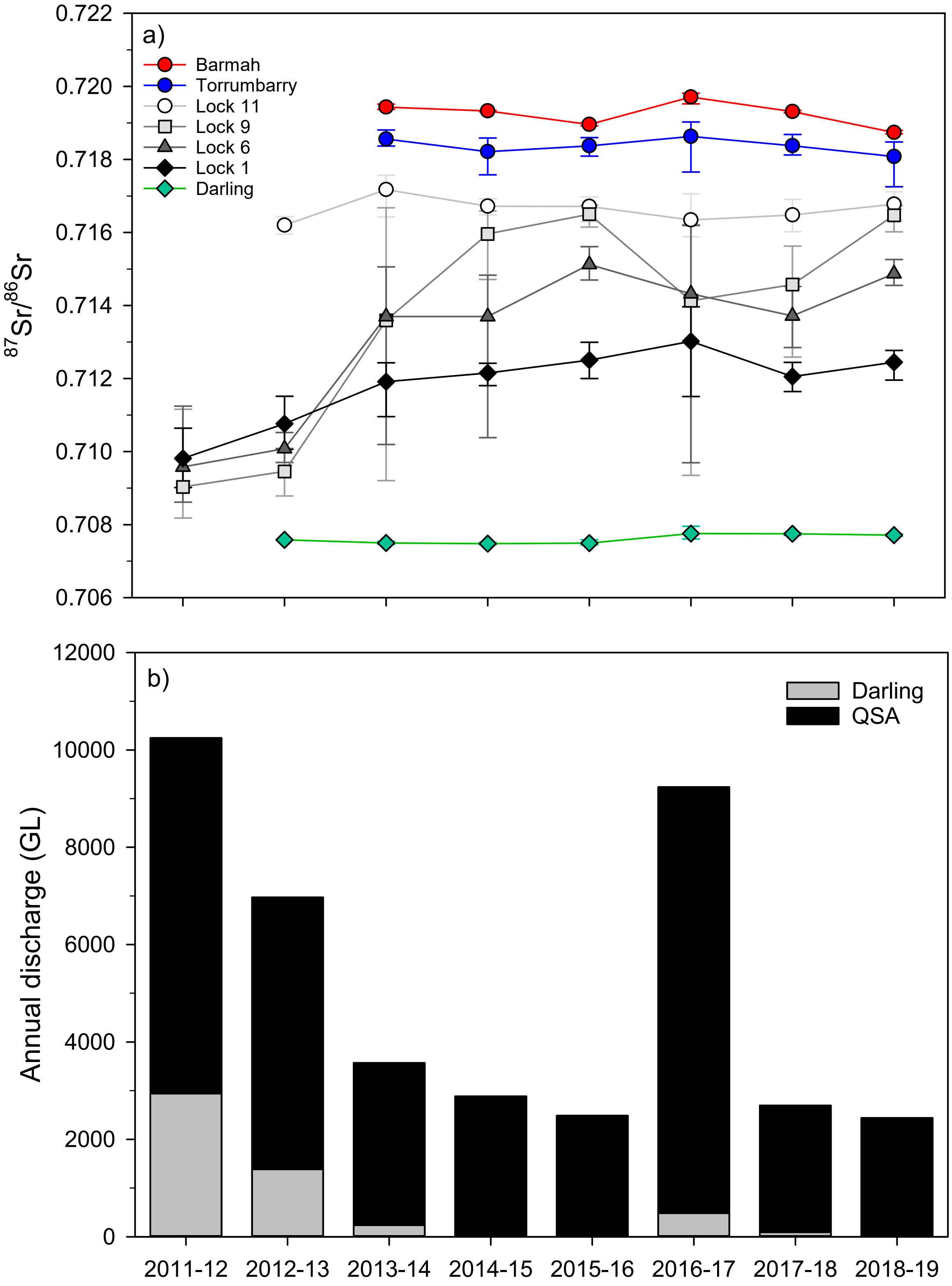

Figure 32. (a) Mean ^{87}Sr/^{86}Sr (with minimum and maximum values as error bars) in water samples collected from spring–summer in the mid-Murray (Barmah, Torrumbarry and Lock 11), lower River Murray (Lock 9, 6 and 1) and Darling rivers from 2011–2019, and (b) annual discharge (GL) in the Murray River at the South Australian border (QSA) and the proportion of discharge from the Darling River at Burtundy that contributed to QSA.

Larvae and eggs

Abundance

Catch-per-unit-effort (CPUE) of golden perch larvae and fish eggs was temporally variable from 2010–2019. Abundance of larvae was relatively high (>8 ind./trip) between 2010-11 and 2013-14, peaking in 2011-12 (77 ± 51 ind./trip) (Figure 33a). In contrast, abundance was consistently low (<2.5 ind./trip) from 2014-15 to 2018-19.

Total egg abundance followed a similar pattern to golden perch larval abundance and was high (>100 ind./trip) from years 2010-11–2013-14 (Figure 33b), and low from 2014-15 to 2018-19 (<30 ind./trip), with the exception of the high flow year of 2016-17 (212 ± 142 ind./trip). Perch eggs (golden or silver perch) were not differentiated from other eggs prior to 2015-16. Nevertheless, total egg abundance is considered indicative of perch egg abundance because they formed the majority (97%) of the sampled pelagic fish egg community from 2015-16–2018-19 (Figure 33b).

Spawn date and origin

Golden perch larvae were collected annually below Locks 1 and 6 in the LMR from 2014–2019, except for 2015-16 when larvae (n = 2) were only collected below Lock 1. Ages of these larvae ranged 2–42 days, corresponding to spawn dates between 26 September and 7 January (Figure 34). Most larvae collected across the five-year period originated from spawning that occurred on ascending and descending limbs of flow pulses (peaks ≥10,000 ML/d) between early November and late December, and in association with water temperatures ≥20°C.

Larvae collected from 2014–2019 predominantly exhibited otolith core $^{87}Sr/^{86}Sr$ comparable to water $^{87}Sr/^{86}Sr$ in the lower River Murray (Murray River downstream of the Darling River junction) (Figure 33). In 2016-17, a 27-day-old larvae collected from Lock 6, born on 14 December 2016, had otolith core $^{87}Sr/^{86}Sr$ indicative of the Darling River. Similarly, in 2017-18, a 7-day larvae collected from Lock 6, born on 27 December 2017 had otolith core $^{87}Sr/^{86}Sr$ indicative of the mid-Murray River above Lock 11.

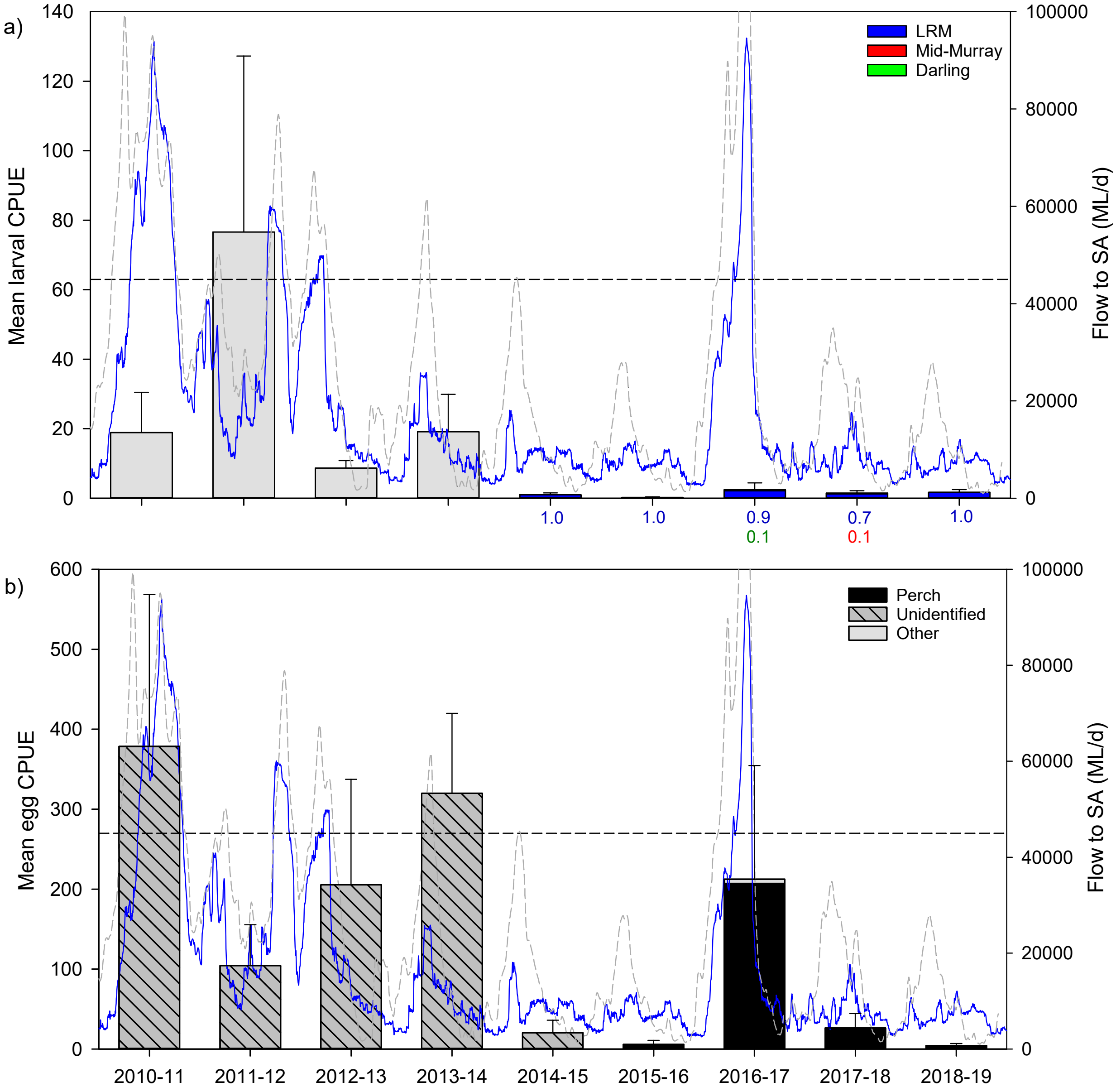

Figure 33. Mean catch-per-unit-effort (CPUE, individuals per trip) ± standard error of (a) golden perch larvae and (b) fish eggs collected below Lock 6 and Lock 1, in the LMR, from 2010-11 to 2018-19 during larval tow sampling. From 2014-15 onwards, natal origins of larvae (expressed in text as proportions) are inferred through a subsample of otolith core $^{87}Sr/^{86}Sr$ from larval and golden perch of that spawn year. LRM = Murray River, below the Darling confluence. Perch (i.e. silver or golden perch) eggs were not differentiated from other eggs prior to 2015-16. Daily flow to South Australia (SA) (solid blue line), modelled flow under non-regulated (natural) conditions (dotted grey line) and approximate bankfull flow (dotted black line) are shown.

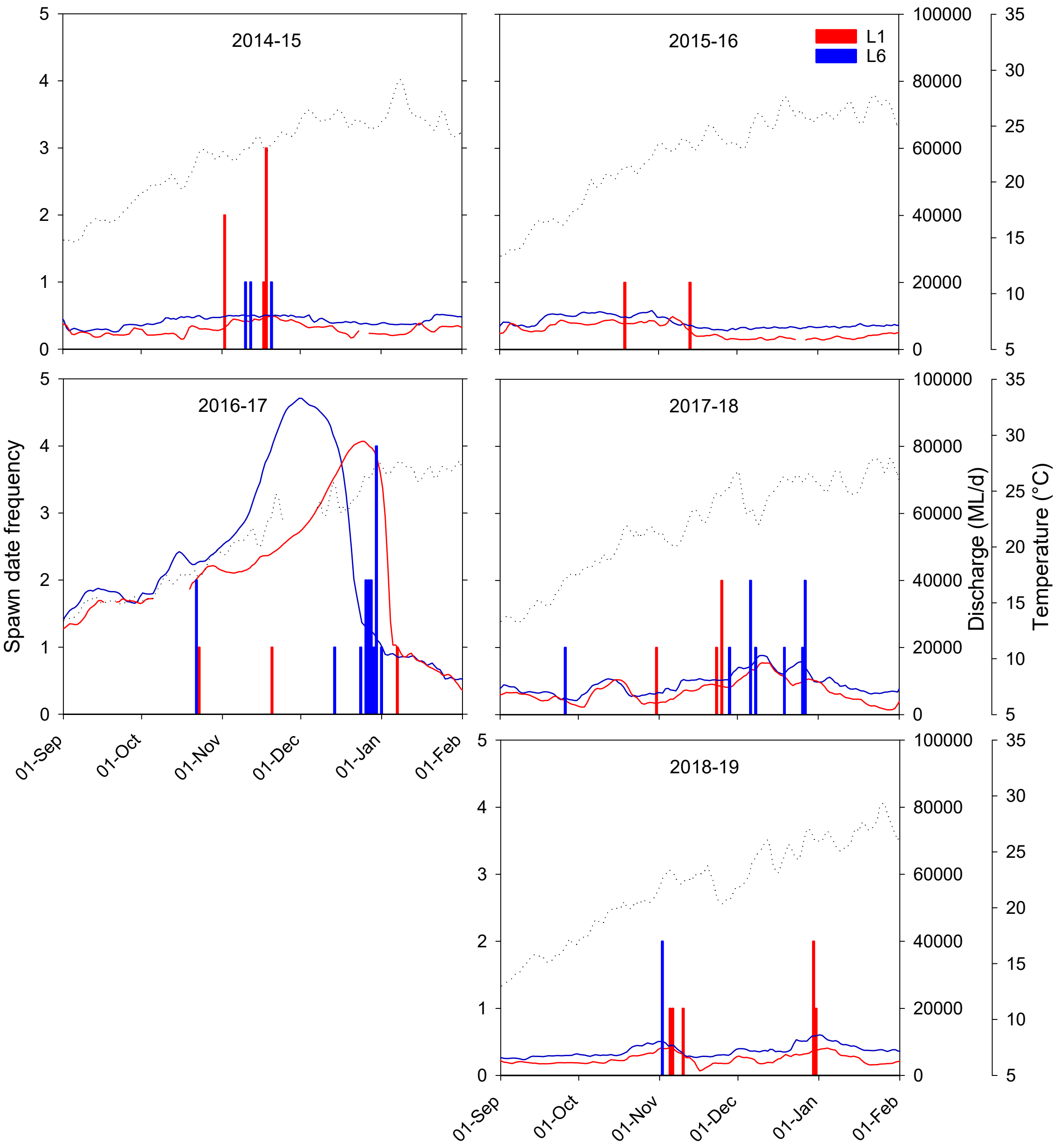

Figure 34. Back-calculated spawn dates (frequency) for larval golden perch captured at Lock 6 (blue bars) and Lock 1 (red bars) in the LMR during spring/summer 2014–2019. Spawn dates are plotted against discharge (ML/d) at the South Australian border (solid blue line) and downstream of Lock 1 (solid red line), and water temperature (°C) (dotted black line).

Juvenile/adult abundance

Relative abundance of golden perch generally declined from 2015 (0.57 ± 0.08 ind./shot) to 2019 (0.28 ± 0.02 ind./shot, Figure 35). This was supported by PERMANOVA which demonstrated significant differences between years (Pseudo $F_{4,44}$=4.0271, *P*=0.0078). Pair-wise comparisons revealed significant differences between 2019 and 2015, and 2019 and 2016, but not between any other years (Table 21).

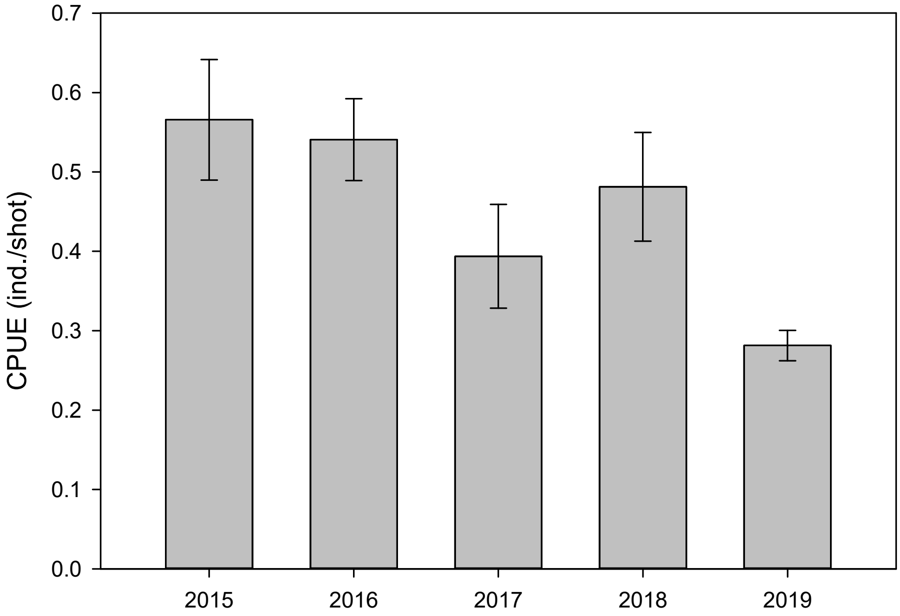

Figure 35. Mean catch-per-unit-effort (CPUE) ± standard error of golden perch captured during Category 1 Fish Assemblage electrofishing (individuals per 90 second shot) in the gorge geomorphic zone (10 sites) of the LMR in autumn from 2015–2019. CPUE data from five sites are presented for 2017 as other sites were sampled during winter 2017.

Table 21. PERMANOVA pairwise comparison test results for golden perch electrofishing abundance (individuals per 90 second shot) in the gorge geomorphic zone (10 sites) of the LMR in autumn from 2015–2019. *P*-values presented in bold are significant comparisons, using Bonferroni corrected α = 0.0050 (Narum 2006) for comparisons between years (ten comparisons).

Comparison	**Large-bodied**	
	t	*P (perm)*
2015 vs. 2016	0.27262	0.8132
2015 vs. 2017	1.4563	0.1768
2015 vs. 2018	0.82529	0.4424
2015 vs. 2019	3.6323	**0.0017**
2016 vs. 2017	1.6998	0.1172
2016 vs. 2018	0.69301	0.5219
2016 vs. 2019	4.7217	**0.0006**
2017 vs. 2018	0.8086	0.4507
2017 vs. 2019	2.1526	0.0646
2018 vs. 2019	2.8114	0.0150

Length and age structure

From 2015–2019, YOY (age 0+) golden perch were absent from electrofishing, although two individuals were collected in fyke nets in 2017. In 2015, the sampled population of golden perch ranged in age from 2+ to 18+ years, with dominant cohorts of age 4+, 5+, 14+ and 18+ fish, born/originating in 2010-11, 2009-10, 2000-01 and 1996-97, respectively (Figure 36). In 2015, age 4+, 5+, 14+ and 18+ fish comprised 30, 35, 9 and 10% of the sampled population in the LMR, respectively. These four cohorts persisted in the population from 2015–2019, but their relative contribution in the population shifted. In 2019, sampled golden perch ranged in age from 1+ to 22+ years, with dominant cohorts of age 5+, 7+, 8+ and 9+ fish, born/originating in 2013-14, 2011-12, 2010-11 and 2009-10, respectively, and comprising 9, 23, 41 and 13% of the sampled population in the LMR, respectively.

Otolith $^{87}Sr/^{86}Sr$, natal origin and migration history of golden perch

From 2015–2019, a total of 206 golden perch (excluding larvae) were analysed for natal origin and migration history, from the following cohorts: 2016-17 (*n* = 6), 2013-14 (*n* = 19), 2012-13 (*n* = 13), 2011-12 (*n* = 34), 2010-11 (*n* = 56), 2009-10 (*n* = 46), 2005-06 (n = 9), 2000-01 (*n* = 8) and 1996-97 (*n* = 15) (Figure 36).

Individuals from the three most dominant year classes, 2011-12, 2010-11 and 2009-10, exhibited otolith core $^{87}Sr/^{86}Sr$ comparable to water $^{87}Sr/^{86}Sr$ in the lower River Murray (Murray River downstream of the Darling River junction) and the Darling River (Figure 36; Figure 32). This indicates that some fish from these cohorts were born and spent their entire lives in the lower River Murray, whilst others originated from the Darling River and transitioned into the lower River Murray in their first (i.e. age 0+, Figure 37a) or second year (i.e. age 1+, Figure 37b), and remained in this region until capture.

Golden perch from the 2005-06 cohort exhibited otolith core $^{87}Sr/^{86}Sr$ comparable to water $^{87}Sr/^{86}Sr$ in the lower River Murray and the mid-Murray River (Figure 36; Figure 32). This indicates that some fish from this cohorts were born and spent their entire lives in the lower River Murray, whilst others originated from the mid-Murray River, upstream of Lock 11, and transitioned into the lower River Murray early in their first year (i.e. age 0+, Figure 37c), and remained in this region until capture.

Dominant cohorts of older fish 2000-01 and 1996-97, generally exhibited otolith core $^{87}Sr/^{86}Sr$ comparable to water $^{87}Sr/^{86}Sr$ in the lower River Murray (Figure 36; Figure 32). Transects indicated these fish had spent their entire lives in the lower River Murray (Figure 37d).

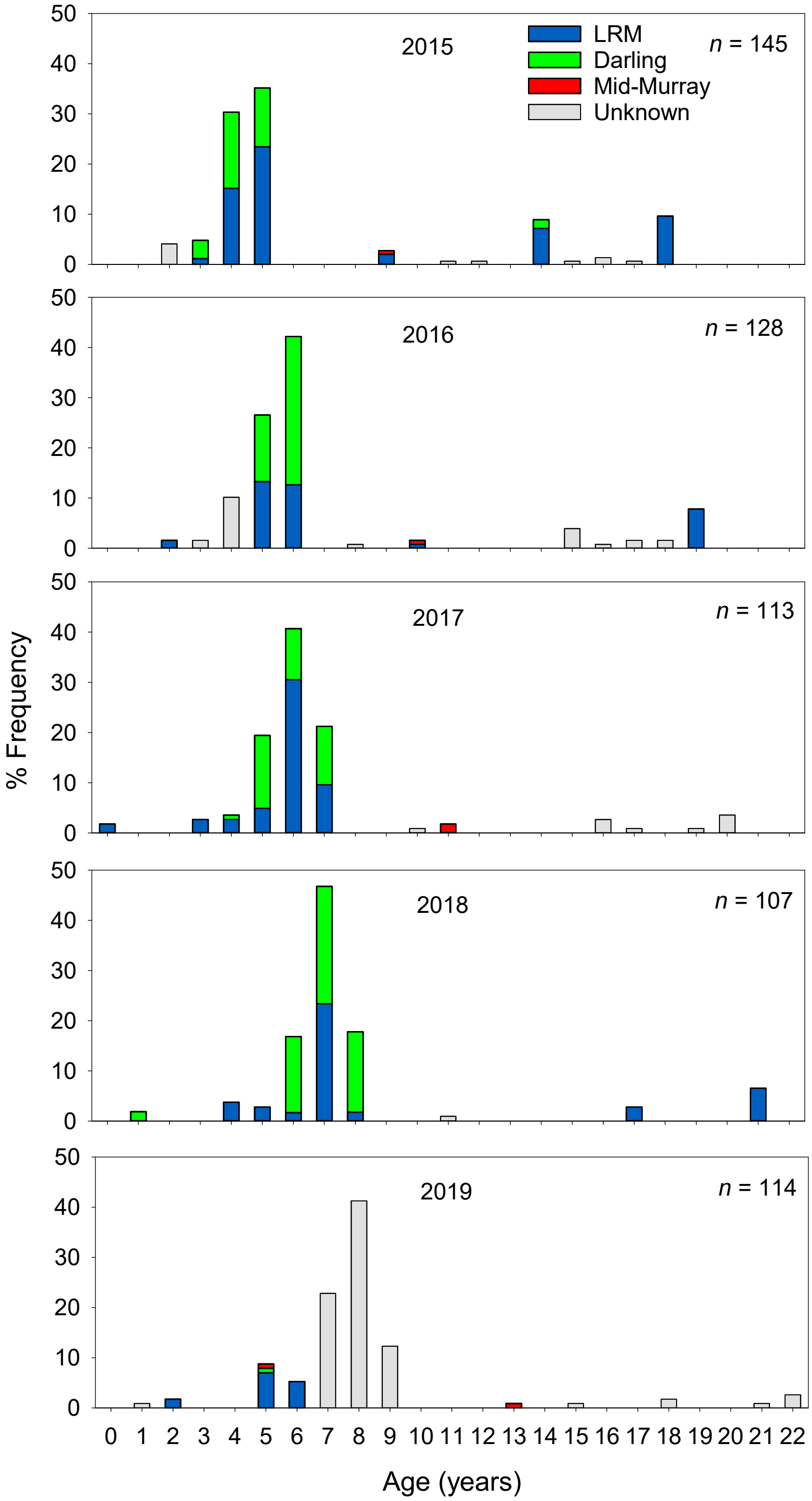

Figure 36. Age frequency distribution of golden perch from the LMR from 2015–2019 showing the natal origins of dominant cohorts inferred from otolith core signatures of the sampled fish in comparison to the water sample reference collection (Figure 32). LRM = Murray River, below the Darling confluence. Percentage of origin for each cohort are based on the subsampled population. Age cohorts with grey bars were not assessed for natal origin.

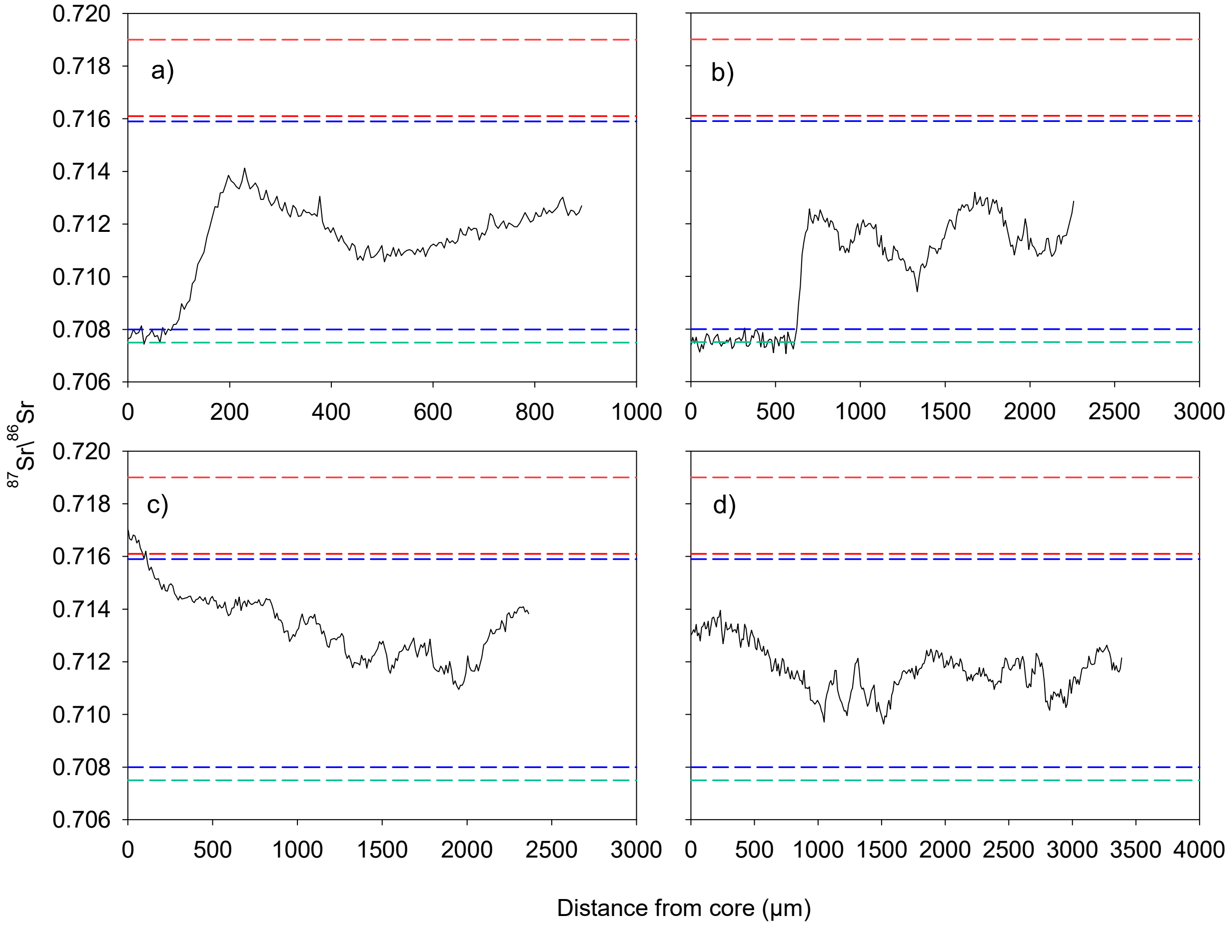

Figure 37. An individual life history profile based on transect analysis of ^{87}Sr/^{86}Sr from the core to edge of an otolith from a (a) 2016-17 year class (age 1+) golden perch from Lowbank, (b) 2010-11 year class (age 8+) golden perch from Cobdogla, (c) 2005-06 year class (age 11+) golden perch from Rilli Island and (d) 1996-97 year class (age 21+) golden perch from Swan Reach. Green dashed line indicates the temporally stable water ^{87}Sr/^{86}Sr of the lower Darling River (i.e. ~0.7075) and the blue dashed lines represent the range of water ^{87}Sr/^{86}Sr in the lower River Murray (i.e. ~0.7080–0.7160). Red dashed lines represent the range of water ^{87}Sr/^{86}Sr in the mid-Murray River (Lock 11–Torrumbarry, ~0.7160–0.7190) (Figure 32).

Evaluation

The evaluation approach, including assessment criteria, is described in the evaluation section for hydrological regime (Section 2.1).

Table 22. Fish spawning and recruitment evaluation questions and answers. CEW = Commonwealth environmental water, YOY = young-of-year.

CEWO evaluation questions	**Outcomes of CEW delivery**				
	2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to reproduction of golden perch?	Coincident spawning, negligible recruitment	Coincident spawning, negligible recruitment	Coincident spawning, negligible recruitment	Coincident spawning, negligible recruitment	Delivery of CEW to the LMR in 2017-18 coincided with spawning, but no detectable recruitment of golden perch (to YOY, age 0+).
What did CEW contribute to the resilience of golden perch?	From 2014-15 to 2018-19, flow (including CEW) in the LMR was not associated with recruitment of golden perch population. In 2019, ages 0+–4+ collectively represented 2.6% of the population.				

Contribution (to what extent CEW contributed towards the outcome, with the significance of the outcome considered):
⬜ Unknown 🟥 Negative ⬜ None/negligible 🟩 Minor 🟩 Moderate 🟦 Substantial

Discussion

Migratory, pelagic broadcast-spawning fishes are disadvantaged by the anthropogenic modification of rivers. Spawning migrations are interrupted by barriers, hydrologic cues for spawning are altered by flow regulation, and the obligate downstream drift of eggs and larvae is interrupted by the physical and hydraulic impacts of dams and weirs (Dudley and Platania 2007; Perkin *et al.* 2015). Many pelagic spawning fishes also demonstrate *periodic* life history strategies (Winemiller and Rose 1992), with high recruitment variability and low demographic resilience.

Golden perch and silver perch are migratory, pelagic spawning fish, native to the MDB. For both species, spawning and recruitment have been associated with hydrological variability and increased water temperatures during spring–summer (Mallen-Cooper and Stuart 2003; Zampatti and Leigh 2013a; 2013b). In the southern MDB, flow regulation, particularly the elimination of a perennial spring flow pulse (Mallen-Cooper and Zampatti 2018), compromises the recruitment and population resilience of golden perch and silver perch (Zampatti and Leigh 2013; Tonkin *et al.* 2019). As such, these species form a focus for environmental water management that aims to improve recruitment and restore population resilience.

Over the five-year term of this project, we aimed to identify potential associations between reproduction (spawning and recruitment) of golden perch and environmental water delivery. Our specific objective was to identify timing of spawning and the spatial source of successful recruits to enable association of ecological response with hydrology; and to explore population connectivity between regions of the southern connected MDB. We hypothesised that (1) increased flow (nominally >15,000 ML/d) in spring–summer would promote the spawning and recruitment to young-of-year (YOY), and (2) multiple years of enhanced spring–summer flow would increase the resilience of golden perch populations in the LMR by promoting a diverse age structure.

From 2014–2019, increased spawning, as indicated by the abundance of eggs and larvae, was associated with elevated in-channel (>20,000 ML/d) and overbank (>45,000 ML/d) flows in concert with water temperatures ≥20°C during spring–summer. The high flow year of 2016-17 was associated with high egg abundance, relative to low flow years (2014-15, 2015-16, 2017-18, 2018-19). Egg abundance was of similar magnitude to other high flow years (i.e. 2010-11 to 2013-14) that were associated with recruitment and distinct cohorts of golden perch in the LMR population contingent. Nevertheless, in 2016-17, larval abundance was low and subsequent recruitment to YOY was negligible. It is possible that a hypoxic blackwater event associated with overbank floods in 2016 (Ye *et al.* 2018), compromised the survival of eggs and larvae, thus leading to recruitment failure.

In 2015, the sampled population of golden perch ranged in age from 2+ to 18+ years, with dominant year classes from 2010-11, 2009-10, 2000-01 and 1996-97. These cohorts persisted in the population to 2019, but the relative proportions of older age cohorts (e.g. 2009-10, 2000-01 and 1996-97) declined. Recruitment from 2015–2019, through either localised spawning or immigration, was poor and did not promote any strong year classes. In association with an ageing population, abundance of golden perch in the LMR steadily declined. Over the same period, declines in golden perch abundance were also observed in other monitoring programs in the region, for example, in *condition* monitoring at the Chowilla icon site (Fredberg *et al.* 2019). Potential factors contributing to decline are a lack of recruitment, mortality (fishing or natural) and upstream emigration of adults (Zampatti *et al.* 2018). Whilst in 2019, the golden perch population in the LMR is primarily comprised of mature adults (i.e. age 5+–9+ years), a lack of younger cohorts and low age structure diversity lead to a population that lacks resilience to environmental perturbations.

Recruitment and the establishment of distinct cohorts of golden perch in the LMR is influenced by local spawning and the downstream transport and migration of larvae/YOY (age 0+) and juveniles (predominantly age 1+) from spatially distinct spawning areas (Zampatti 2019). Fish originating from years of high flow (i.e. 2009-10, 2010-11 and 2011-12) in the Darling River contributed substantially to prominent age cohorts in the LMR. Thus, spawning and recruitment in the Darling River, and physical and hydrological connectivity between the Darling and Murray rivers, are substantial drivers of golden perch population dynamics of the lower River Murray. Whilst some downstream transport of eggs, larvae and early-stage juveniles from the Darling River to the lower River Murray may occur during low-intermediate flows, it appears that bank-full–overbank flows in both the Darling and Murray rivers are associated with significant downstream dispersal of juvenile golden perch from recruitment sources in the Darling River (Zampatti and Leigh 2013a; Zampatti 2019).

Management implications

From 2014-19, >500 GL/year of Commonwealth environmental water were delivered to the LMR for a range of ecological objectives, including increased spawning and recruitment of flow-dependent species. These allocations generally increased discharge at the South Australian border from base (entitlement) flows (~3,000–7,000 ML/d) to small freshes (~12,000–17,800 ML/d) during spring/summer. For golden perch, some spawning coincided with periods when environmental water was used to promote flow pulses in the LMR. Nevertheless, recruitment was negligible. Evidence from this and allied investigations suggest that greater flow rates (>20,000 ML/d) are likely required to significantly influence golden perch spawning and recruitment in the LMR. Perennial spring–summer in-channel flow pulses of this magnitude were a key feature of the hydrograph of the LMR prior to regulation (Mallen-Cooper and Zampatti 2018), and would have occurred in recent years under natural, unregulated conditions (Figure 5).

Ultimately, fish behaviour and life history processes are influenced by the hydraulic characteristics of flow (i.e. hydrodynamics), rather than volumes of water and rates of flow (hydrology). Riverine hydrodynamics likely provide cues for movement and spawning, and facilitate the downstream dispersal of early life stages. Contemporary flow management in the MDB mainly focuses on volume and discharge, whereas a hydraulic perspective is more relevant to understanding ecological processes and eliciting ecological outcomes (Bice *et al.* 2017; Mallen-Cooper and Zampatti 2018). This is particularly relevant in the LMR, where contiguous weir pools result in a highly modified hydraulic environment. Indeed, the river only begins to substantially regain its lotic character at flows >20,000 ML/d (Bice *et al.* 2017). Decreasing weir height reinstates lotic characteristics at lower discharges and, in conjunction with restoration of spring pulses, may improve the recruitment and population structure of riverine fishes such as golden perch. A key knowledge gap, however, remains the role that water velocity and turbulence plays in the suspension, transport and retention of golden perch eggs and larvae.

Conclusion

There has been no substantial recruitment of golden perch in the LMR since 2012-13, leading to a population dominated by a few distinct and ageing cohorts. To improve the resilience of golden perch populations in the LMR, it would be pertinent in the coming years to provide flows in the LMR that may facilitate golden perch spawning and recruitment. Specifically, Commonwealth environmental water could contribute to spring/early summer in-channel flow >20,000 ML/d.

2.6 Fish Assemblage

Background

From 2015–2019, we collected fish assemblage data in the main channel of the LMR to inform Basin-scale evaluation of fish community responses to Commonwealth environmental water. This evaluation is being undertaken by the Centre for Freshwater Ecosystems at La Trobe University.

Objectives

In this report, our objectives are to: (1) provide summary statistics of the catch rates and population demographics for nominated species; (2) describe temporal variation in fish assemblage and population structure from 2015–2019; and (3) discuss key findings based on published research and our current understanding of fish life histories and population dynamics in the LMR. Our interpretations of the data for this indicator do not infer association with Commonwealth environmental water delivery.

Methods

During autumn 2015–2019, small- and large-bodied fish assemblages were sampled from the gorge geomorphic zone of the LMR (Figure 1) using fyke nets and electrofishing, respectively. Prescribed methods (Hale *et al.* 2014) were used to obtain population structure data for seven target species (Figure 38). Refer to SARDI *et al.* (2018) for detailed sampling design and methodology. In 2017, electrofishing at half of the sites (sites 1, 3, 4, 6, 7) was delayed to winter due to equipment failure/malfunction.

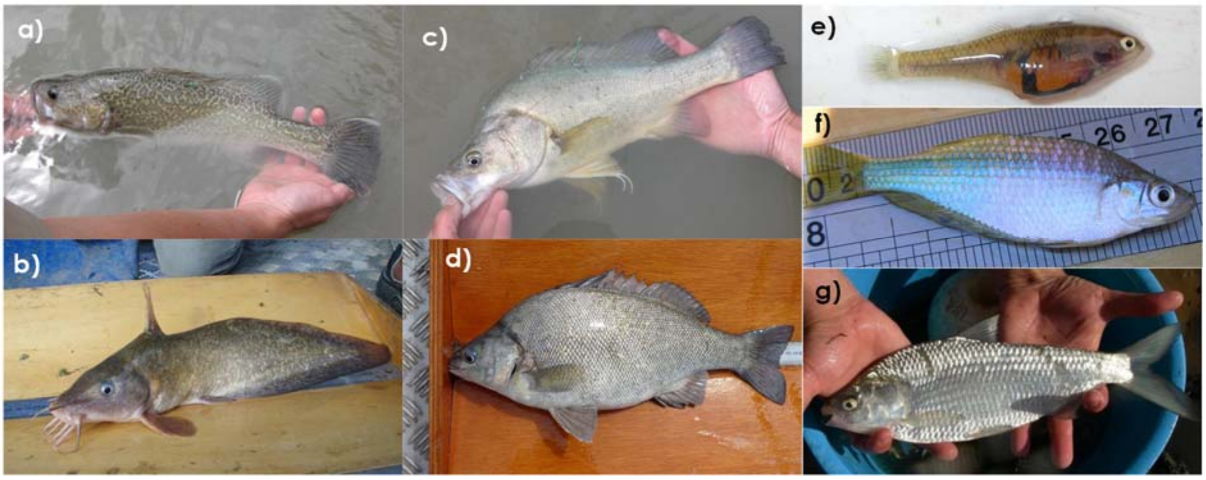

Figure 38. Target species for the LMR: (a) Murray cod and (b) freshwater catfish (equilibrium life history); (c) golden perch and (d) silver perch (periodic life history); and (e) carp gudgeon, (f) Murray rainbowfish and (g) bony herring (opportunistic life history).

Temporal variation in fish assemblage structure (species composition and abundance), between sampling years, was investigated using Non-metric Multi-Dimensional Scaling (MDS), permutational multivariate analysis of variance (PERMANOVA) and Similarity Percentages (SIMPER) analysis in the software package PRIMER v. 6.1.12 (Clarke and Gorley 2006) and PERMANOVA + v.1.02 (Anderson *et al.* 2008). To determine temporal variation in population structure, length frequency histograms were qualitatively compared between sampling years.

Results

Catch summary

From 2015–2019, a total of 43,010 individuals (ind.) from eight large-bodied fish species were collected by electrofishing. Across all years, bony herring (*Nematalosa erebi*) was consistently the most abundant species (75–97% of the catch/annum), followed by common carp (*Cyprinus carpio*), whose proportional abundance of the total catch varied across sampling years (1% in 2015 to 20% in 2017) (Figure 39a).

A total of 112,028 individuals from seven small-bodied species were collected by fyke nets. The total abundance of small-bodied species varied considerably across sampling years (9,661 ind. in 2017 to 37,678 ind. in 2018), but the proportional abundance of individual species remained similar. Carp gudgeon (*Hypseleotris* spp.) was the most abundant species (86–92% of catch/annum), followed by Gambusia (*Gambusia holbrooki*) (6–9%/annum) (Figure 39b).

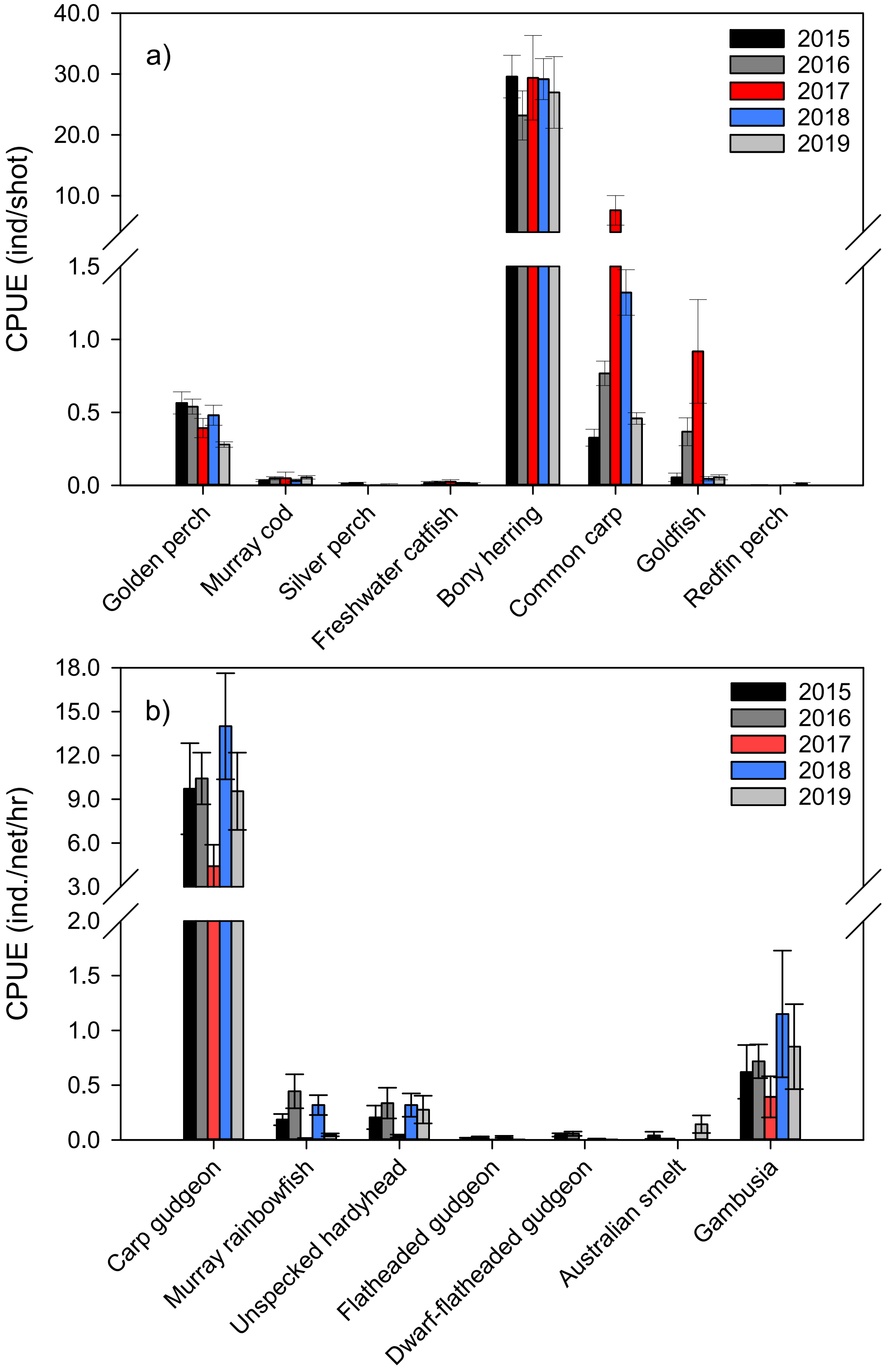

Figure 39. Mean catch-per-unit-effort (CPUE) ± standard error of (a) large-bodied fish species captured using electrofishing (individuals per 90 second shot) and (b) small-bodied fish species captured using fine-mesh fyke nets (individuals per net per hour) in the gorge geomorphic zone (10 sites) of the LMR in autumn from 2015–2019. Electrofishing CPUE data from five sites are presented for 2017 as other sites were sampled during winter 2017.

Temporal variability in fish assemblage structure

MDS ordination of electrofishing data demonstrated separation of 2017 samples from all other years (Figure 40a). PERMANOVA indicated that large-bodied fish assemblages were significantly different between years (*Pseudo*-$F_{4,44}$ =5.0917, $P \leq 0.001$). Pairwise comparisons revealed significant differences between 2017 and all other years, but not between any other year comparisons (Table 23).

Table 23. PERMANOVA pairwise comparison test results for large- and small-bodied fish assemblages in the gorge geomorphic zone of the LMR from autumn 2015–2019. *P*-values presented in bold are significant comparisons, using Bonferroni corrected α = 0.0050 (Narum 2006) for comparisons between years (ten comparisons).

Comparison	**Large-bodied**		**Small-bodied**	
	t	*P (perm)*	*t*	*P (perm)*
2015 vs. 2016	2.0305	0.0068	1.1526	0.2412
2015 vs. 2017	3.5839	**0.0008**	2.0765	0.0131
2015 vs. 2018	1.7638	0.0382	1.207	0.2167
2015 vs. 2019	1.2874	0.1733	1.0302	0.3327
2016 vs. 2017	2.2942	**0.0035**	3.7883	**0.0001**
2016 vs. 2018	1.7945	0.0276	0.81273	0.5858
2016 vs. 2019	1.6992	0.0441	1.7025	0.0461
2017 vs. 2018	2.6821	**0.0013**	3.1583	**0.001**
2017 vs. 2019	3.1843	**0.0004**	1.9555	0.02
2018 vs. 2019	1.7619	0.0417	1.4932	0.0955

For small-bodied fish assemblages, there were significant differences between years (*Pseudo*-$F_{4,49}$ =3.8453, $P \leq 0.001$). PERMANOVA pair-wise comparisons revealed significant differences in small-bodied fish assemblages between 2017 and 2016, and 2017 and 2018, but not between any other year comparisons (Table 23).

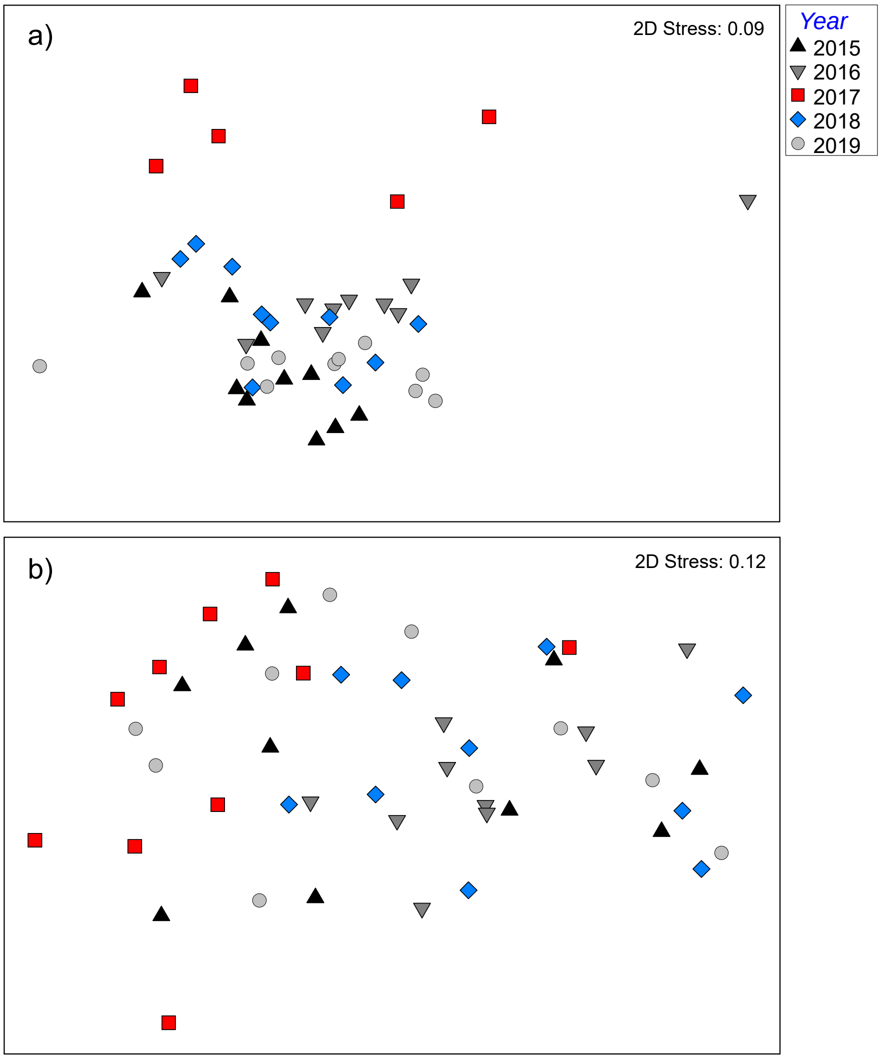

Figure 40. Non-metric multi-dimensional scaling (MDS) plot of (a) large-bodied fish assemblages sampled by electrofishing and (b) small-bodied fish assemblages sampled by fyke netting in the gorge geomorphic zone of the LMR from 2015–2018. Sites (*n* = 5) sampled in winter 2017 were removed from the ordination.

SIMPER indicated that differences between years for large-bodied fish assemblages were primarily driven by higher abundance of common carp in 2017, lower abundance of common carp in 2015 and 2019, and lower abundance of bony herring in 2016 (Figure 39). SIMPER indicated that differences between 2017 and 2016 and 2018 for small-bodied fish assemblages were driven by a lower relative abundance of carp gudgeon in 2017 (Figure 39).

Temporal variation in length/age structure of large-bodied species

In 2015, the sampled golden perch population was mostly comprised of age 4+ (35%), 5+ (25%) and 18+ (13%) cohorts. In 2018, these cohorts persisted as age 7+ (51%), 8+ (19%) and 21+ (4%), respectively, but age 6+ fish were also apparent (23%) (Figure 41).

Low numbers of silver perch (*Bidyanus bidyanus*) and freshwater catfish (*Tandanus tandanus*) were sampled from 2015–2019. In 2015, silver perch ranged in age from 2+ to 5+ years, whilst only one silver perch (age 1+) was sampled in 2018 (Figure 42). Freshwater catfish ranged in age from 5+ to 9+ years in 2015, and 8+ to 13+ years in 2018 (Figure 43).

In 2015, the sampled Murray cod population was represented by individuals 103–145 mm (age 0+) and 1310 mm (not sacrificed for ageing). In 2018, the sampled Murray cod population consisted of individuals 74–140 mm (age 0+), 307 mm (age 1+), 409 mm (not sacrificed, potentially age 2+ or 3+) and 515 mm (not sacrificed, potentially age 3+ or 4+) (Figure 44).

From 2016 to 2019, the sampled bony herring population ranged in age from 0+ to 6+ years and was dominated by age 0+ fish (83–91%) in each year (Figure 45).

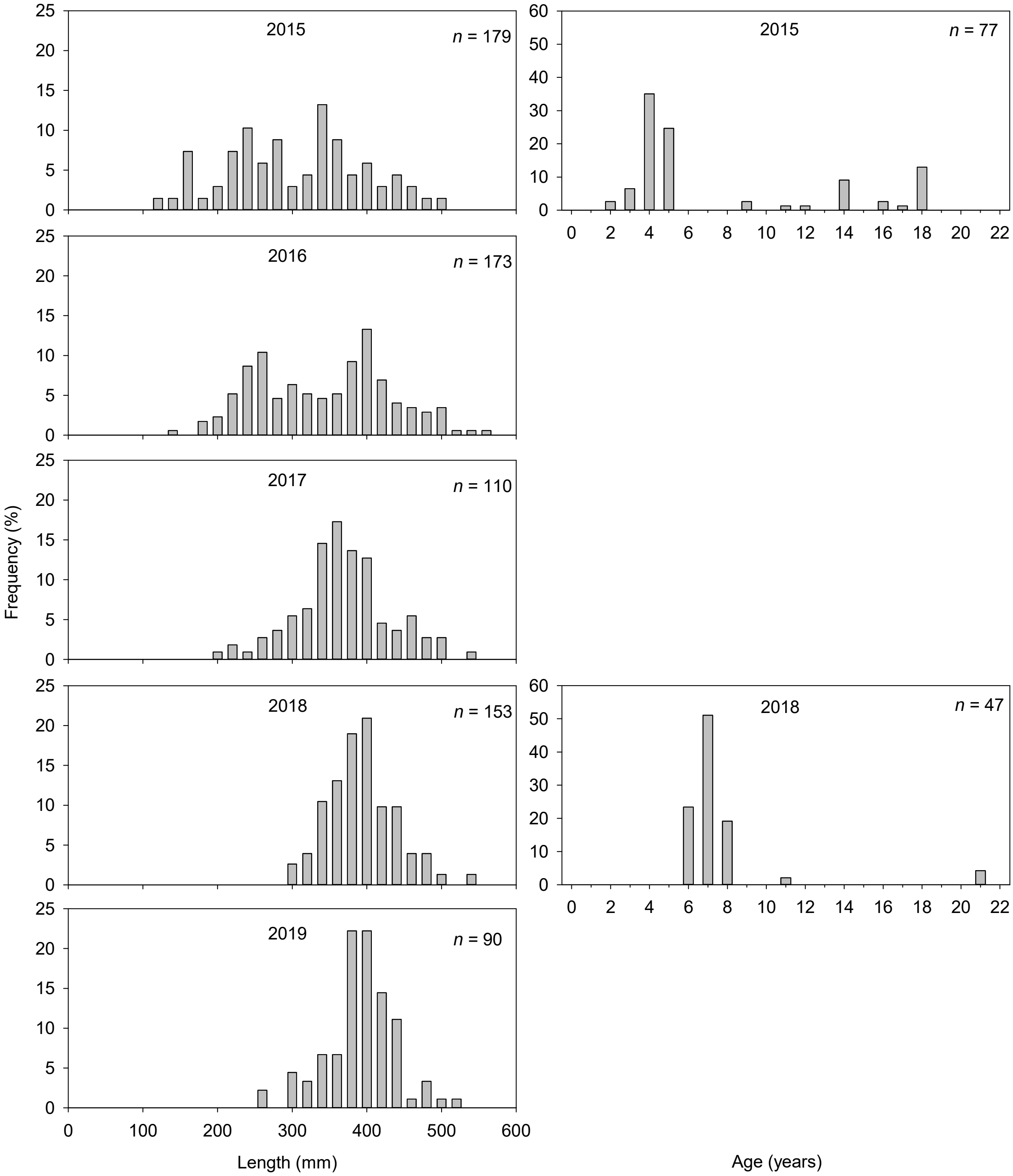

Figure 41. Length frequency distributions and age structures of golden perch collected from the gorge geomorphic zone of the LMR from 2015–2019.

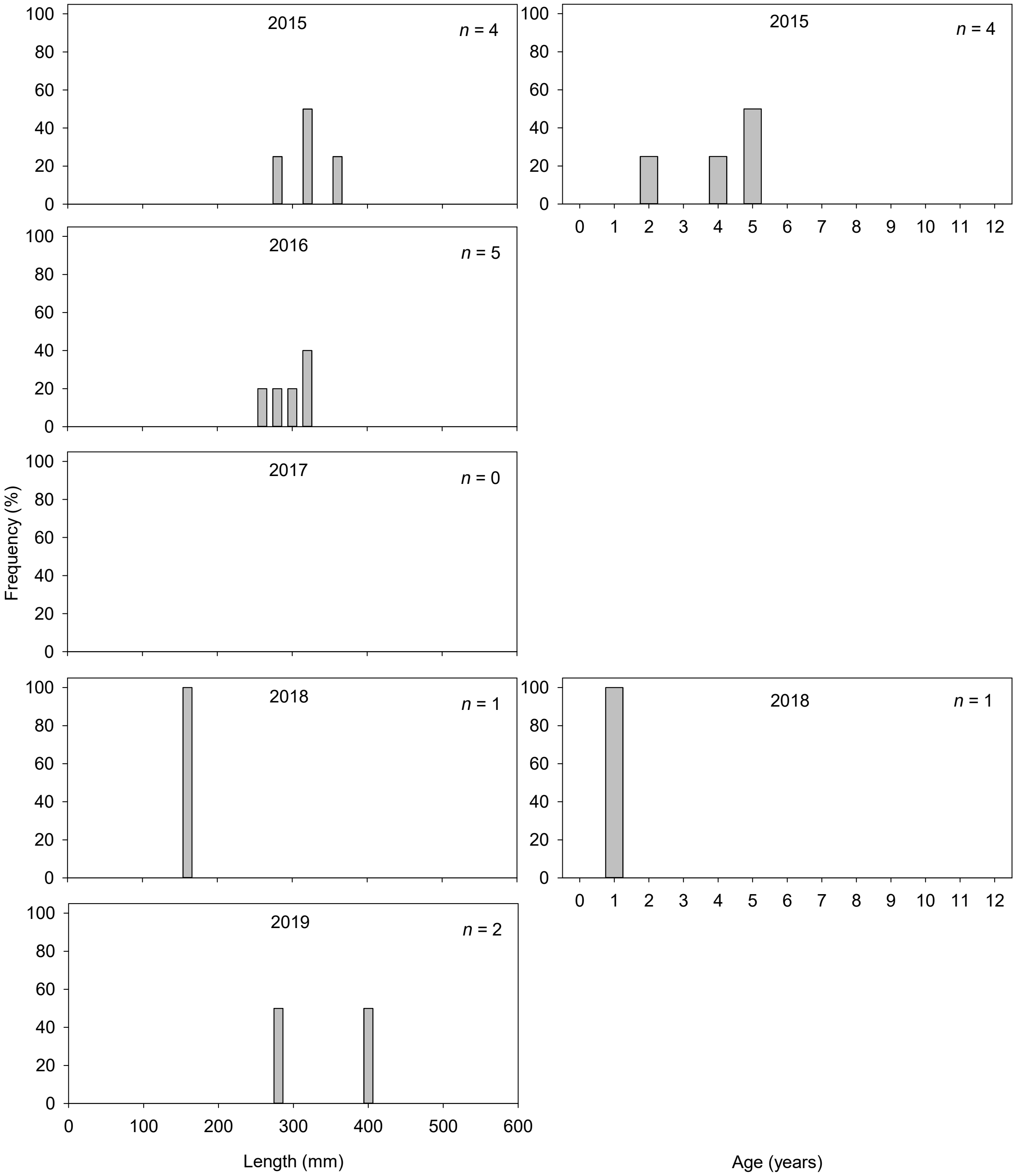

Figure 42. Length frequency distributions and age structures of silver perch collected from the gorge geomorphic zone of the LMR from 2015–2019.

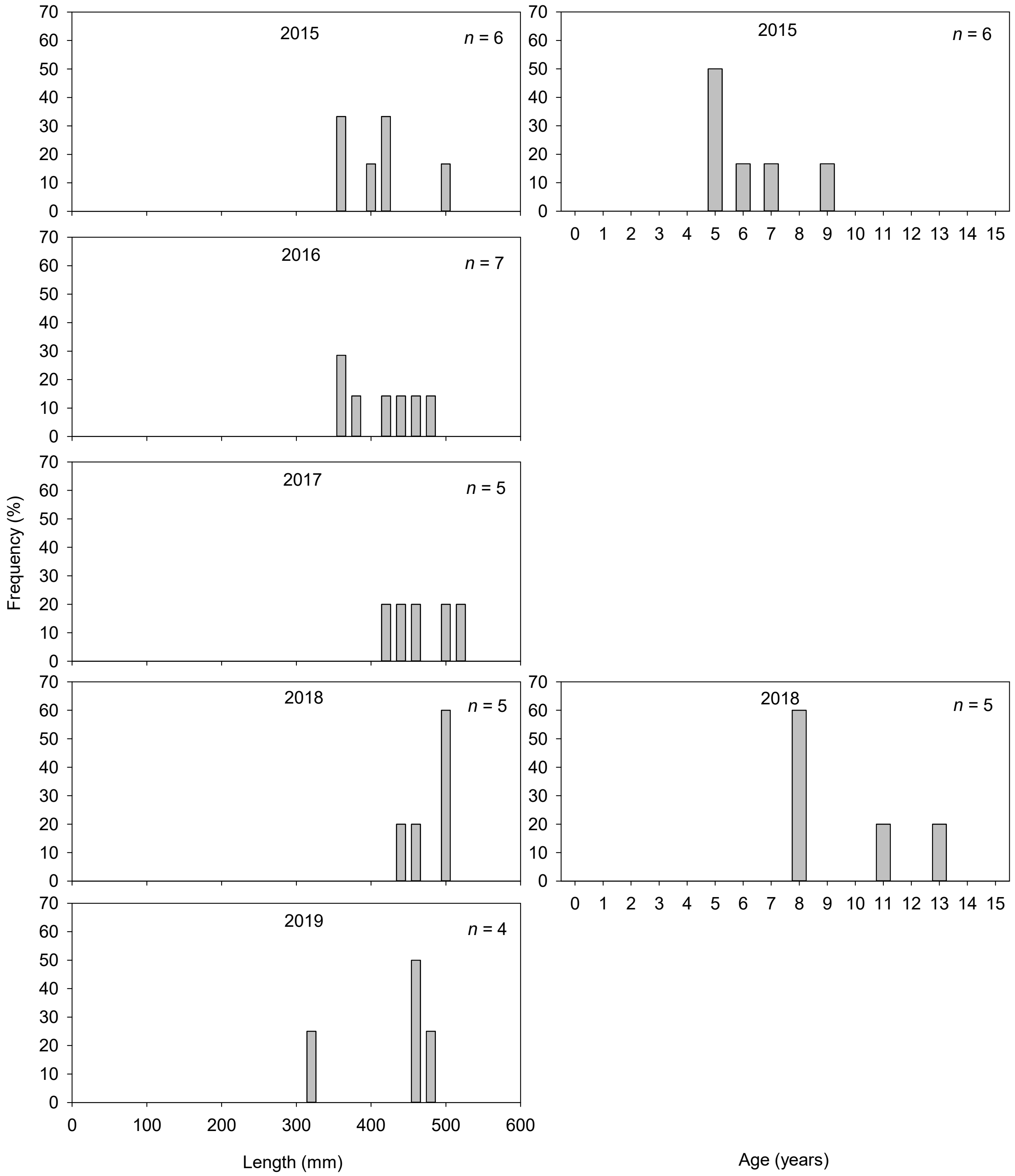

Figure 43. Length frequency distributions and age structures of freshwater catfish collected from the gorge geomorphic zone of the LMR from 2015–2019.

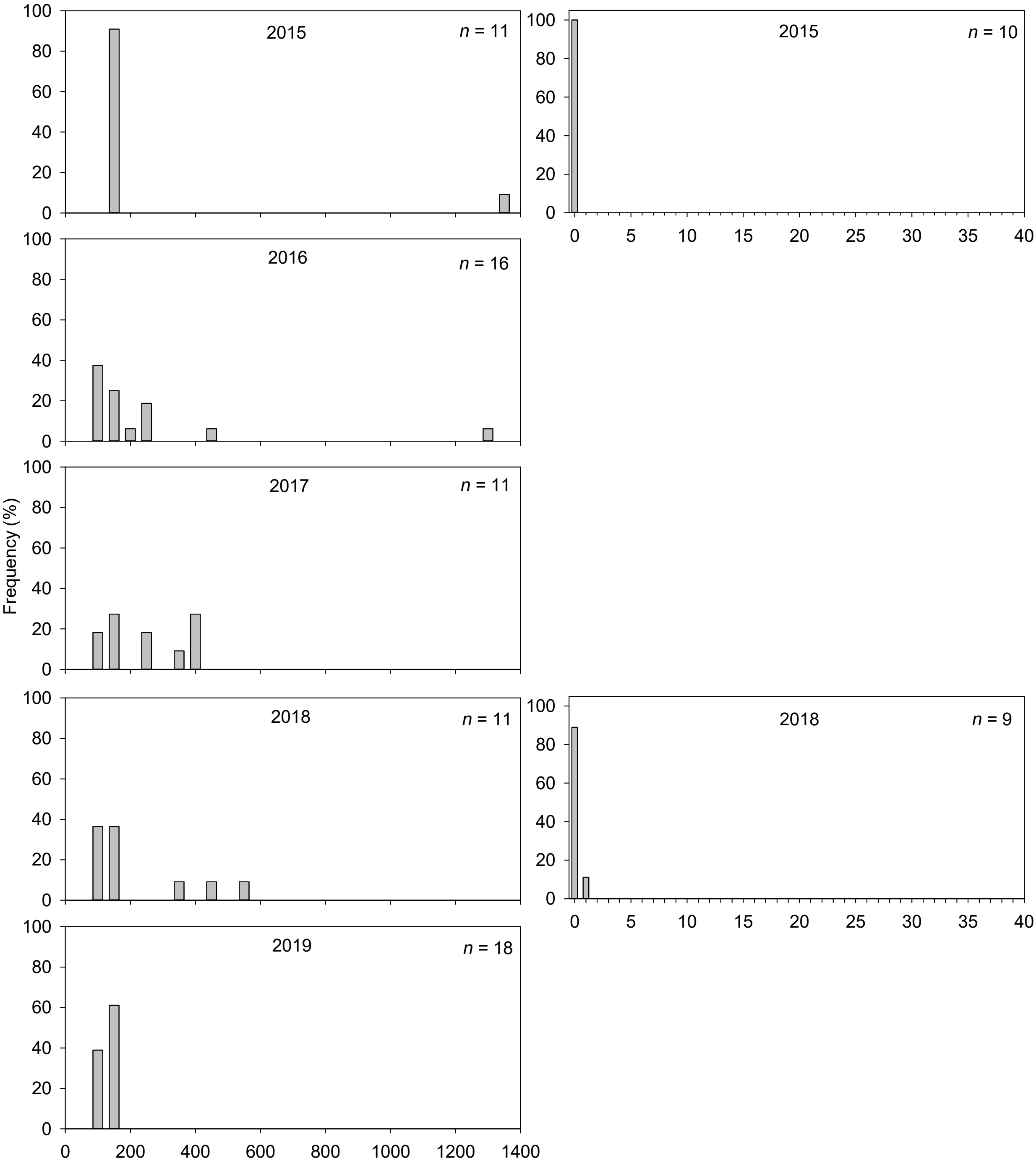

Figure 44. Length frequency distributions and age structures of Murray cod collected from the gorge geomorphic zone of the LMR from 2015–2019.

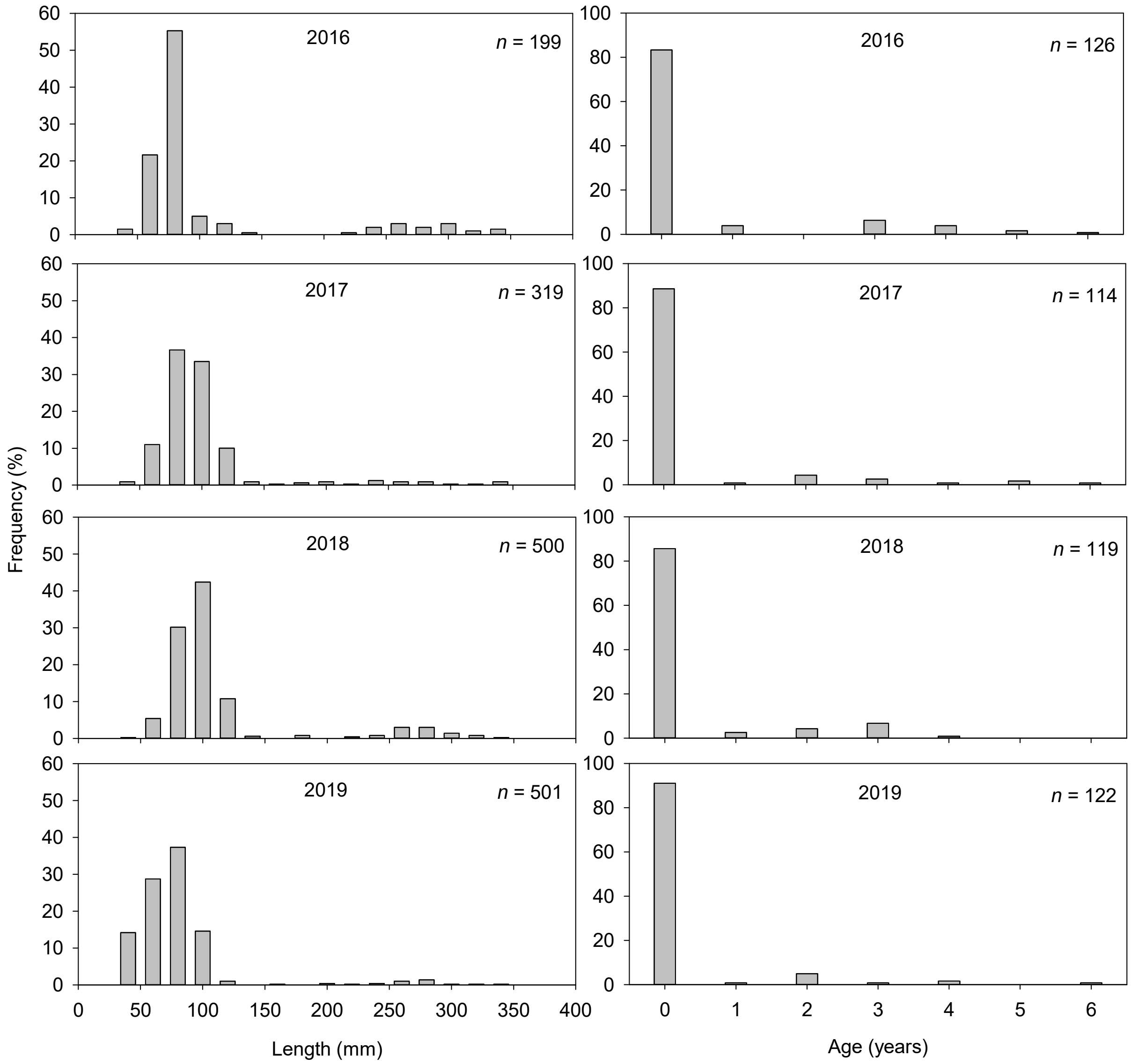

Figure 45. Length frequency distributions and age structures of bony herring collected from the gorge geomorphic zone of the LMR from 2016–2019.

Evaluation

There are no CEWO evaluation questions for this indicator for the LMR Selected Area. The Basin-scale evaluation of fish community responses to Commonwealth environmental water is being undertaken by the Centre for Freshwater Ecosystems at La Trobe University. For this report, fish monitoring data were consolidated to evaluate a number of fish targets of DEW's Long-Term Watering Plan (Appendix C).

Discussion

During 2014-15 and 2015-16, relatively low (<15,000 ML/d), stable flows predominated in the LMR. In these years, small-bodied fish abundance and diversity were high. Abundances of flow-cued spawning species (i.e. golden perch and silver perch) remained similar in both years and overall, fish assemblage structure was characteristic of low flows in the LMR and similar to that during drought in 2007–2010 (Bice *et al.* 2014).

In 2017, following flooding in spring–summer 2016 (peak flow ~94,600 ML/d), there was a significant change to the small- and large-bodied fish assemblages, with an overall decrease in the abundances of small-bodied species and an increase in the abundance of common carp. A reduction in submerged vegetation in the main channel of the LMR during 2016-17, due to a combination of increased water depth/decreased light penetration and physical scour, likely resulted in a decrease in habitat availability and decreased abundance of small-bodied fishes (Bice *et al.* 2014). In 2017, increased abundance of common carp appeared to be driven by a large recruitment event in 2016-17 associated with flooding. Following a recession in water levels in summer 2017, large numbers of age 0+ common carp likely entered the main channel from off-channel floodplain and wetland habitats (their typical spawning habitat) and were captured during sampling in autumn and winter 2017.

The fish assemblage in 2017 was more typical of high flows, similar to the one in 2010–2012 (Bice *et al.* 2014). Nevertheless, in 2016-17, recruitment of native, large-bodied flow-cued spawners (e.g. golden perch) was negligible, despite a flow regime that was potentially conducive to spawning of these species (Mallen-Cooper and Stuart 2003; Zampatti and Leigh 2013a; 2013b) (also see Section 2.5). It is possible, that water hypoxia, associated with blackwater during the spring–early summer spawning season (Ye *et al.* 2018), impacted the survival of eggs and larvae.

Following in-channel flows (up to 17,800 ML/d) in spring–early summer 2017-18, small-bodied fish species composition and abundance reverted back to that of pre-flood conditions (i.e. 2016 and 2015), presumably due to structural and hydraulic habitats (i.e. submerged vegetation and stable water levels) conducive to small-bodied fish recruitment in the main river channel. The large-bodied fish assemblage trended towards one typical of 'low flows' (e.g. 2016, 2015 and 2008, Bice *et al.* 2014) due to a reduction of common carp abundance in 2018, relative to 2017. Common carp abundance, however, was still significantly greater in 2018, relative to 2016 and 2015, indicating the progression of fish from the 2016-17 cohort (age 0+) into the population as age 1+ (Ye *et al.* 2019). In 2019, following similar hydrological conditions to 2018 (i.e. low, within channel flows <18,000 ML/d), the small- and large-bodied fish assemblages remained comparable to the previous year. Common carp abundance, however, continued to decline in 2019 to levels similar to 2015 and 2016.

Based on electrofishing length frequency data, no recruitment (to age 0+) was observed for freshwater catfish in the LMR from 2014–2019. In the LMR, the spawning biomass of freshwater catfish may be historically low (Ye *et al.* 2015b) and their recruitment dynamics are poorly understood.

From 2015–2019, regular recruitment of Murray cod (i.e. fish <150 mm TL) was observed in the LMR, including during years following an in-channel flow pulse (15,000–18,000 ML/d) and a high, overbank flow (>90,000 ML/d), but also during three years of low, stable, in-channel flows (<12,000 ML/d). Furthermore, these cohorts have generally persisted in the population, although were not sampled during 2019, likely due to low abundance and/or inadequate sampling effort. These results contrast with data collected from 2003–2010, during the Millennium Drought, when Murray cod recruitment, measured as abundance of YOY in autumn, was limited in the predominantly lentic main channel habitats of the LMR. Subsequently, recruitment was observed in association with spawning that occurred in high flow years from 2010–2013 (Zampatti *et al.* 2014). The mechanisms that facilitate recruitment of Murray cod (to YOY) in the LMR are unclear, but likely relate to enhanced survival of early life stages associated with riverine hydraulics and productivity. The CEWO Monitoring, Evaluation and Research (MER) Project aims to explore these mechanisms by assessing aspects of Murray cod recruitment (e.g. abundance, growth, condition) in association with flow.

Management implications

Prolonged low, in-channel flows (<15,000 ML/d) promote hydraulic (e.g. lentic) and structural (submerged plant) habitat conditions suitable for low flow generalist species, e.g. small-bodied species. Increased variability in the annual flow regime that includes large, overbank (>45,000 ML/d) flows will lead to increased recruitment of a range of species with different life histories (e.g. flow-dependant), promoting diversity and resilience in the fish assemblage. These desired flow outcomes may not be achieved with environmental water due to large volumes of water required and/or operational constraints. Nevertheless, coordinated environmental water delivery could contribute to within-channel flows ~20,000 ML/d and thereby promote variability in the annual flow regime, including hydraulics.

Conclusion

During 2015 and 2016, in the main channel of the LMR, fish assemblages were characterised by high abundances of small-bodied species and a lack of recruitment of native, large-bodied flow-cued spawners. This fish assemblage structure was similar to that during drought in 2007–2010 (Bice *et al.* 2014) and characteristic of a low flow scenario. Following high flows in 2016-17, the fish assemblage shifted towards one characterised by low abundances of small-bodied species and high abundance of a large-bodied species, common carp. This assemblage was more typical of high flows, similar to 2010–2012 (Bice *et al.* 2014). Nevertheless, recruitment of native, large-bodied flow-cued spawners (e.g. golden perch) was negligible in 2016-17, likely due to water hypoxia associated with the blackwater event. During 2018 and 2019, a return to low, in-channel flows (<18,000 ML/d) resulted in the fish assemblage trending back towards that of 2015 and 2016, following: an increase in small-bodied fish abundance; a lack of recruitment from native, flow-cued spawners; and a decrease in common carp abundance.

3 SYNTHESIS AND EVALUATION

Investigations into the ecological responses to Commonwealth environmental water in the LMR from 2014–2019 enabled us to address a series of evaluation questions for CEWO, which were adapted from Basin-scale questions to be relevant for this Selected Area (Table 24; SARDI *et al.* 2018). Additional evaluation questions from the Department for Environment and Water (DEW), which relate to ecological targets of the South Australian Murray River Long-Term Environmental Watering Plan (LTWP), are discussed in Appendix C. The contribution of environmental water to hydraulic condition and matter transport was assessed throughout each year using a modelling approach, whereas other indicators of ecological responses (stream metabolism, microinvertebrates and fish spawning and recruitment) were assessed through empirical monitoring.

The last five-year period (2014-15–2018-19) was dominated by low flow conditions (i.e. flow <18,000 ML/d at the South Australian border) except for 2016-17. Commonwealth environmental water contributed to baseflows and freshes in the LMR, particularly as winter and spring–early summer flow pulses (Figure 5), mainly via return flows from upstream watering events. Environmental water helped to increase river flows up to 10,000, 11,700, 17,800 and 12,100 ML/d during spring/summer in 2014-15, 2015-16, 2017-18 and 2018-19, respectively. In these dry years, flows in the LMR would have been minimal (~5,000–7,000 ML/d) without environmental water. In 2016-17, with high, unregulated overbank flows during spring/early summer (peak ~94,600 ML/d), environmental water delivery mostly occurred after mid-December and assisted in slowing and extending the flood recession in summer.

Commonwealth environmental water also facilitated the weir pool manipulations at Locks 2 and 5 (both lowering and raising) in 2015-16, 2017-18 and 2018-19, and helped support other infrastructure operation (e.g. Chowilla regulator events, Appendix A). Furthermore, Commonwealth environmental water supplemented freshwater flows to the Lower Lakes and maintained barrage releases to the Coorong for 9–12 months each year. Environmental water delivery, in combination with infrastructure operation, contributed to a number of ecological outcomes in the LMR Selected Area. Key outcomes are summarised below, whilst a summary of the evaluated outcomes are presented in Table 24. Refer to each indicator evaluation section (Section 2) for a more detailed interpretation and criteria for assessment.

Environmental water delivery in most periods of the last five years increased longitudinal connectivity through the system. Noting that annual discharge volume only provides a coarse representation of longitudinal connectivity, in 2015-16 and 2017-18, with >35% increases in annual discharge to the LMR due to environmental water, the Basin-wide environmental watering strategy target of >30% flow increase in the Murray River (here calculated at the SA border) was achieved. During these years, lateral connectivity also improved with increased inundation of the littoral zone of the river channel, predominately via weir pool raisings, supported by Commonwealth environmental water.

Environmental water increased water velocities and introduced some hydraulic variability in the LMR that would not have otherwise occurred over the last five years (excluding unregulated flow in 2014-15 and 2016-17). This was better demonstrated in 2017-18, when an extra 49 km (14%) of river was transformed to lotic (flowing water, velocities >0.3 m/s) conditions (for at least 14 days, or 36 km (10%) of river for at least 30 days due to the Commonwealth environmental water. This event was associated with a greater flow increase to ~18,000 ML/d, compared to other dry years. The timeframe of ~14 days may correspond with more rapid, flow-related ecological processes or biological responses such as mobilisation of carbon and nutrients, microinvertebrate reproduction and community turn over, or fish egg/larval drift and dispersal, whereas ~30 days, for example, generally coincides with the time period from spawning to the completion of larval development for golden perch (Lake 1967; Tonkin *et al.* 2006). This implies potential benefits of environmental water to the LMR ecosystem, although the spatial/temporal scale of effect may be limited, even in 2017-18. Improving riverine hydraulics is critical for ecological restoration in the LMR. Many native biota that have life history strategies adapted to a flowing river are currently extinct or have suffered major declines due to the largely weir pool environment in this region (Mallen-Cooper and Zampatti 2018). Pre-regulation, the LMR was characterised by lotic, riverine habitats, with water velocities ranging ~0.2–0.5 m/s, even at discharges <10,000 ML/d (Bice *et al.* 2017).

Environmental water, in combination with weir pool raisings, also increased water level variability in the LMR each year. Periodic increases in water levels can improve the condition of littoral vegetation (Gehrig *et al.* 2016) and increase biofilm diversity (Steinman and McIntire 1990), which is a key component of riverine food webs.

In most years, environmental flows made a significant contribution, particularly during spring–summer periods, to reducing the risk of low dissolved oxygen (DO) in the LMR, by increasing water mixing (velocities >~0.2 m/s) and oxygen exchange at the surface. The potential low oxygen period was reduced for >2 months in 2015-16 and 2017-18, and >1 month in the other years. This is critical as spring–summer period corresponds with high ecosystem respirations rates and is the primary reproductive season of many species that generally favour DO >5 mg/L. The consequences of low oxygen on survival of aquatic biota are evident from the flood year in 2016-17, when dissolved oxygen levels fell to 0 mg/L in the LMR for a short period and resulted in a depletion of larger microinvertebrates (e.g. cladocerans and copepods) and extensive kills of Murray cod.

Environmental flows contributed to small increases in river production in the LMR, as indicated by 1–2% increases in cross-sectional GPP at LK6ds, due to the largely fixed water levels set by weirs. Modelling for the Murray River at Hattah demonstrated that the potential influence of environmental flows on riverine production, thus ‘carry capacity’, can be substantial in un-regulated sections of the Murray River.

The microinvertebrate assemblage variation across the four years (2014–2018) was associated with variability in flow (which included environmental water) to the LMR. The dominance of pelagic rotifers characterised the assemblages in low flow years (2014-15 and 2015-16), whereas more littoral and pelagic organisms played a role in driving assemblage structure during higher flow periods (e.g. 2016-17 overbank flows and 2017-18 flow pulse of ~18,000 ML/d). Environmental water deliveries enhanced connectivity, as indicated by microinvertebrate dispersal and transport from upstream sources (e.g. from Goulburn/Upper Murray River in 2017-18, from northern tributaries (e.g. Darling River) in 2016-17) to the LMR. Dispersal of microinvertebrates contributed to increased diversity and potentially provided a more diverse food source for higher trophic organisms. Increases in species richness and density were evident during different periods in all years, coinciding with increases in discharge to the LMR (in some cases in conjunction with weir pool raising and/or the operation of the Chowilla regulator), supported by Commonwealth environmental water. Lower trophic level diversity has been found to be linked to enhanced productivity and community resilience of microinvertebrates (e.g. Striebel *et al.* 2012) as well as enhanced diversity in higher trophic organisms (e.g. Eadie and Keast 1984).

From 2014–2019, some spawning of golden perch occurred in the LMR, coinciding with small in-channel freshes (~12,000–18,000 ML/d) promoted by Commonwealth environmental water, and unregulated overbank flows (>45,000 ML/d) during spring–summer. However, no YOY (age 0+) were detected over last five years during electrofishing, indicating localised recruitment failure and low levels of immigration from spatially distinct spawning sources such as the lower Darling and mid Murray rivers. The golden perch population is currently dominated by a few distinct and ageing cohorts in the LMR. The LTIM and allied studies suggest that flows (>20,000 ML/d) in the LMR or substantial flow pulses (e.g. 2,000–3,000 ML/d) in the lower Darling River that reach the LMR (Zampatti and Leigh 2013a; Zampatti *et al.* 2015) during spring–summer are likely required to significantly influence golden perch recruitment in the LMR. Such hydrological characteristics were present in 2016-17, and golden perch eggs were sampled in much greater abundances than during the four dry years. The mechanisms leading to recruitment failure in 2016-17 remain unexplored, but may in part be associated with the extensive hypoxic blackwater (Ye *et al.* 2018), which may have impacted directly on egg/larval survival and development, or indirectly via the effect of food resources in the LMR.

Perennial spring–summer in-channel flow pulses of >20,000 ML/d were a key feature of the flow regime in the LMR prior to regulation (Mallen-Cooper and Zampatti 2018), and would have occurred in recent years under natural, unregulated conditions (Figure 5). Under current conditions, the river only begins to substantially regain its lotic (flowing water) character at flows >20,000 ML/d (Figure 12; also see Bice *et al.* 2017). Decreasing weir height can also help restore lotic characteristics at lower discharges (e.g. Bonifacio *et al.* 2016). Riverine hydrodynamics are critical for life history processes of flow-cue spawning fish species, including providing cues for movement and spawning, and facilitating the downstream dispersal of early life stages. They are also a key driver for riverine ecological functions (e.g. productivity, matter transport, food web). Reinstating lotic habitats is particularly relevant for ecological restoration in the LMR, where contiguous weir pools result in a highly modified hydraulic environment. Specifically, environmental water could contribute to spring/early summer flow pulses >20,000 ML/d, in conjunction with weir pool lowering, which may improve the recruitment and population structure and resilience of riverine fishes such as golden perch.

Over the last five years, Commonwealth environmental water played a critical role in maintaining continuous barrage flows and connectivity between the river and the Coorong estuary to support a functioning river system. There was substantially increased salt export out of the Basin, reduced salt import into the Coorong, and reduced salinity levels in the Coorong, improving estuarine habitats and creating favourable conditions for estuarine species. Environmental flows also increased transport of nutrients and phytoplankton, which would likely have stimulated primary and secondary productivity in downstream ecosystems, providing potential benefits to food webs of the LMR, Lower Lakes, Coorong and Southern Ocean, adjacent to the Murray Mouth.

Table 24. CEWO evaluation questions by indicators for the Lower Murray River (LMR) and the Coorong, Lower Lakes and Murray Mouth (CLLMM). Evaluation questions are sourced or adapted from Gawne *et al.* (2014). Evaluation of CEW for hydraulic and matter transport questions is based on modelled data. CEW = Commonwealth environmental water. Refer to the evaluation in respective indicator sections (Section 2) for more detail.

CEWO evaluation questions	**Outcomes of CEW delivery**				
	2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to hydraulic diversity within weir pools? (LMR)	▨	▨	▨	🟩	▨
What did CEW contribute to variability in water levels within weir pools? (LMR)	▨	🟩	▨	🟩	▨
What did CEW contribute to hydrological connectivity (lateral and longitudinal)? (LMR)	▨	🟩	⬜	🟩	▨
What did CEW contribute to dissolved oxygen levels? (LMR)	⬜	🟦	🟩	🟦	🟩
What did CEW contribute to patterns and rates of primary productivity? (LMR)	⬜	⬜	⬜	⬜	⬜
What did CEW contribute to patterns and rates of decomposition? (LMR)	⬜	▨	▨	▨	▨
What did CEW contribute to salinity levels and transport? (LMR, CLLMM)	🟩	🟩	▨	🟩	▨
What did CEW contribute to the salinity regime? (LMR, CLLMM)	🟩				
What did CEW contribute to nitrogen and silica concentrations and transport? (CLLMM)	▒	▒	▒	▒	▒
What did CEW contribute to phosphorus concentrations and transport? (CLLMM)	⬜	▨	⬜	▨	▨
What did CEW contribute to phytoplankton concentrations? (CLLMM)	⬜	⬜	⬜	⬜	⬜
What did CEW contribute to phytoplankton transport? (CLLMM)	⬜	⬜	▨	▨	⬜
What did CEW contribute to water quality to support aquatic biota and normal biogeochemical processes? (LMR, CLLMM)	🟦				
What did CEW contribute to ecosystem function? (LMR, Lakes)	▨				
What did CEW contribute to ecosystem function? (Coorong)	🟦				
What did CEW contribute to microinvertebrate diversity? (LMR)	🟩	▨	▒	▨	▒
What did CEW contribute via upstream connectivity to microinvertebrate communities? (LMR)	🟩	🟩	▒	🟩	▒
What did CEW contribute to microinvertebrate density? (LMR)	▨	▨	▒	⬜	▒
What did CEW contribute to the timing and presence of key species in relation to the diet of large-bodied native fish larvae? (LMR)	▒	▒	▒	▒	▒
What did CEW contribute to reproduction of golden perch? (LMR)	⬜	⬜	⬜	⬜	⬜
What did CEW contribute to the resilience of golden perch? (LMR)	⬜				

Contribution (to what extent CEW contributed towards the outcome, with the significance of the outcome considered). See respective indicator evaluation sections (Section 2) for selection criteria.

▒ Unknown 🟥 Negative ⬜ None/negligible ▨ Minor 🟩 Moderate 🟦 Substantial

4 GENERAL MANAGEMENT RECOMMENDATIONS

To restore riverine ecosystems, environmental water has been used to re-establish key features of the natural flow regime in the MDB (MDBA 2012; Koehn *et al.* 2014; Gawne *et al.* 2014; Webb *et al.* 2017), including the significant ecological asset of the main channel of the Murray River (MDBC 2006). In the LMR, this has involved adding to base flows (~South Australian entitlement flows) and promoting or increasing the magnitude, duration and/or frequency of freshes (in-channel flow pulses). Over the long-term, this is expected to make a significant contribution to achieving ecological outcomes in the LMR, through restoring ecological processes and improving habitat for biota in the main channel and associated floodplain/wetlands.

General recommendations for flow management in the LMR are provided below, based on monitoring outcomes from the LTIM Project, in conjunction with our contemporary understanding of flow-related ecology in the LMR. More specific management considerations are provided in Section 2, based on ecological outcomes and findings from indicators. Our findings and recommendations on flow management are most relevant to the spring–summer period as this was the primary period for biological response of selected indicators in the LMR.

In the LMR, environmental water can be used to increase flow variability, e.g. promote in-channel flow pulses. Spring–early summer in-channel flow pulses were key features of the natural hydrograph in the LMR, which are conspicuously absent from the contemporary flow regime. These flow pulses increase longitudinal connectivity and contribute to a broad range of ecological outcomes in riverine and estuarine ecosystems (e.g. increased matter transport, lotic habitats and spawning and migratory cues for riverine fishes). As demonstrated during the LTIM period, particularly in 2017-18, in-channel flow pulses can be generated in the LMR via return environmental flows through coordinated watering events across the southern connected Basin. However, higher flow rates (>20,000 ML/d) may be required to substantially reinstate flowing river characteristics to achieve greater ecological outcomes (e.g. improving riverine production and recruitment of flow-cued spawning fishes) in the LMR. With existing volumes of environmental water and delivery constraints, during dry years, reaching and sustaining flows >20,000 ML/d in the LMR is largely reliant on coordinating flow deliveries across the southern connected Basin, including flows from tributaries (e.g. Goulburn, Murrumbidgee, Darling rivers). Under wetter scenarios, flows >20,000 ML/d may be achieved by delivering environmental water with unregulated flows.

Improving riverine hydraulics (e.g. water velocity and turbulence) is fundamental for ecological restoration in the LMR. Flows of 20,000–45,000 ML/d can significantly improve hydraulic conditions, by transforming >50% of a weir pool from lentic (slower flowing water, median velocities ≤0.3 m/s) to lotic habitat (faster flowing water, >0.3 m/s) (Ye *et al.* 2018). Restoring such hydrodynamic conditions will underpin riverine ecological processes and support the rehabilitation of many declining biota that are adapted to a flowing environment in the LMR. In addition, infrastructure management, such as weir pool lowering, could be considered to complement flows to achieve hydraulic restoration (Figure 12).

Environmental water contributing to flows >45,000 ML/d (above bankfull level) will increase inundation area considerably along the LMR, supporting off-channel processes and floodplain biota (e.g. floodplain vegetation and tree health). In addition, infrastructure management actions, such as Chowilla regulator operation, have been implemented to complement flows to achieve floodplain ecological outcomes (Appendix B). Overbank flow is also an integral part of the natural flow regime in maintaining ecosystem health of floodplain rivers.

The timing of flow delivery is important and should continue to align with ecological objectives and consider biological processes and life history requirements. For example, flow pulses during the spring/summer reproductive season are required to promote spawning and recruitment of flow-cued species; winter/spring flow pluses are needed to facilitate spawning migration of diadromous fish (e.g. lamprey); and summer/autumn flows are critical to reduce salinities and maintain water levels in the Coorong. In the LMR, as a large proportion of environmental water is delivered as return flows, a coordinated approach for environmental water planning and management across the southern Basin is essential. This includes aligning the timing of water delivery to meet biological requirements in this region in order to achieve multi-site ecological outcomes.

Overall, environmental water delivery that promotes longitudinal and lateral connectivity will enhance the productivity in the LMR through increased carbon and nutrient inputs, and matter transport. Water delivery in conjunction with more natural water level changes are desirable to improve instream productivity, although further research will be required to identify favourable water level regimes. Longitudinal connectivity of river flow is also important for the transport and dispersal of aquatic biota (e.g. microinvertebrates, fish larvae) to and throughout the LMR. This study demonstrated that transportation of microinvertebrate taxa, facilitated by environmental flow delivery, from upstream catchments (e.g. Goulburn, upper Murray, and Darling rivers) contributed to the diverse community in the LMR.

Also important is the source of water (i.e. origin). Because water quality (e.g. turbidity, DOC, the amount and form of nutrients) and biological constituents (e.g. plankton, fish larvae) may vary between different sources of water, flows from different upstream sources can influence ecological outcomes in the LMR.

Furthermore, maintaining flow integrity from upstream (e.g. Darling River or mid-Murray) to the LMR is important to support broad-scale ecological processes and promote positive outcomes (e.g. improved productivity, enhanced spawning and recruitment of flow-dependent fishes). In this regard, consideration needs to include: (1) maintaining hydrological integrity (i.e. magnitude, variability and source) of flow from upstream; and (2) the potential effects on water quality and biological attributes by river operations that re-route (e.g. through floodplains or wetlands) or fragment the flow (e.g. by diversions or water storages), which could lead to changes in ecological response and the structure and function of aquatic food webs.

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6 APPENDICES

APPENDIX A: EXPECTED OUTCOMES OF COMMONWEALTH ENVIRONMENTAL WATER IN THE LMR, LOWER LAKES AND COORONG

Table A1. Expected outcomes of Commonwealth environmental watering actions at the catchment (asset) scale for the LMR main channel, Lower Lakes and Coorong (source, CEWO).

LMR channel	Lower Lakes and Coorong
• Maintain species diversity, extend distributions and improve breeding success and numbers of native fish species through: o Flowing habitat and lateral connectivity o Food resources for fish recruitment o Body condition of mature fish o Barrage fishway connectivity for seasonal fish movement o Coorong salinity (habitat) for fish o Critical habitat and water quality; • Maintain the extent and condition of riparian and in-channel vegetation: o Growth of fringing vegetation o Protect extent of *Ruppia* o Lower Lakes littoral vegetation diversity, condition and extent o Extent and condition of inundation dependent trees and non-woody vegetation in low-lying floodplains; • Maintain species diversity, extend distributions and improve breeding success and numbers of water dependent bird species through: o Habitat and food resources o Breeding events; • Contribute to riverine functioning by: o Supporting primary and secondary production along the Murray River through the mobilisation and transport of nutrients, carbon cycling and biotic dispersal; o Contributing to the transport and export of salt and nutrients through the Murray Mouth.	Coorong: • In dry conditions (e.g. 2018-19, 2017-18, 2015-16, 2014-15): maximise estuarine habitat by prolonging barrage releases to support water levels and improve water quality in the North Lagoon for: o Migratory wader food resources o Fish habitat and movement o Reduce salinity peak for *Ruppia*; • Increase barrage flow (September to December) to support recruitment of Ruppia and spawning of estuarine fish (e.g. black bream); Lower Lakes: • Salt export; • Water quality (for consumptive); • Water levels (Basin Plan objectives); • Fringing vegetation health; • Habitat provision (fish, frogs, colonial waterbirds).

APPENDIX B: OVERVIEW OF OTHER WATERING AND MANAGEMENT ACTIVITIES FROM 2014–2019

In addition to environmental water deliveries to the LMR for 2014–2019 (Section 1.4), the following management actions are relevant to the analyses and interpretations in this report.

Other watering and management activities in the LMR

Manipulation of water levels in weir pools

In the LMR, water levels in weir pools have been raised and lowered annually from 2014–2019, particularly Weir Pools 2 and 5 (Table B1; Figure B1). Annual raising events have generally commenced in August, with peak water levels reached in September, and most water levels returning to normal pool level by mid-November. Weir Pools 2 and 5 were lowered in winter 2017, whilst Weir Pool 6 was lowered in winter 2018.

During 2015-16, 2017-18 and 2018-19, Commonwealth environmental water supported weir pool manipulations (both lowering and raising) by accounting for losses (e.g. evaporation). During 2016-17, weir pools were re-filled by unregulated flow following weir pool manipulations. However, it was Commonwealth environmental water underwriting the requirement of the weir-pools re-filling that enabled the river operators to undertake the manipulations.

Refer to LTIM annual technical reports and other reports (e.g. DEWNR 2014; Hanisch *et al.* 2017), etc. for more detail regarding the objectives, timing and magnitude of the events.

Table B1. Weir pool manipulation timing and magnitude details in the LMR from 2014–2019. *indicates years when Commonwealth environmental water accounted for losses.

Year	Weir pool
	1
2014-15	+0.5 m NPL. Raising between late August and mid-November 2014. Water levels returned to normal pool levels by mid-December 2014, and fell an additional 0.1 m below normal pool level.

Year	Weir pool		
	2	5	6
2014-15	+0.5 m NPL. Raising between late August and mid-November 2014. Water levels returned to normal pool levels by mid-December 2014.		+0.4 m NPL In conjunction with the regulator operation between late September and mid-November 2014.
2015-16*	+0.5 m NPL. Raising between early September and early November 2015. Peak water levels in October, returning to NPL by early December 2015.	+0.45 m NPL. Raising between early September and early November 2015. Peak water levels in October, returning to NPL by early December 2015.	

Year	Weir pool		
	2	5	6
2016-17	+0.75 m NPL. Raising occurred between early July and early October 2016, before undergoing a rapid recession in mid-October 2016 to allow for increasing flows and avoid any threat to the structural integrity of the weirs	+0.48 m NPL. Raising occurred between early July and early October 2016, before undergoing a rapid recession in mid-October 2016 to allow for increasing flows and avoid any threat to the structural integrity of the weirs	+0.59 m NPL In conjunction with the regulator operation (Table B2). Raising began in early August with recession began shortly after peak levels were reached, in mid-October 2016.
2017-18*	-0.08 m NPL. Water levels lowered during late July 2017. +0.5 m NPL. Raising between early August and early September 2017. Peak levels through September before undergoing a drawdown in early October back to NPL by mid-October.	-0.08 m NPL. Water levels lowered during late July 2017. +0.45 m NPL. Raising between early August and early September 2017. Peak levels through September before undergoing a drawdown in early October back to NPL by mid-October.	-0.08 m NPL. Water levels lowered during late July 2017. -0.16m NPL. Water levels lowered from early May to late June 2018.
2018-19*	+0.5 m NPL. Raising between early August and early September 2018. Peak levels through September and early October before undergoing a drawdown in mid October back to NPL by mid-November.	+0.35 m NPL. Raising between mid- and late August 2018. Peak levels through September and early October before undergoing a drawdown in mid October back to NPL by early November.	+0.20 m NPL. Raising between mid- August and late September 2018. Peak levels through October before undergoing a drawdown in late October back to NPL by early November.

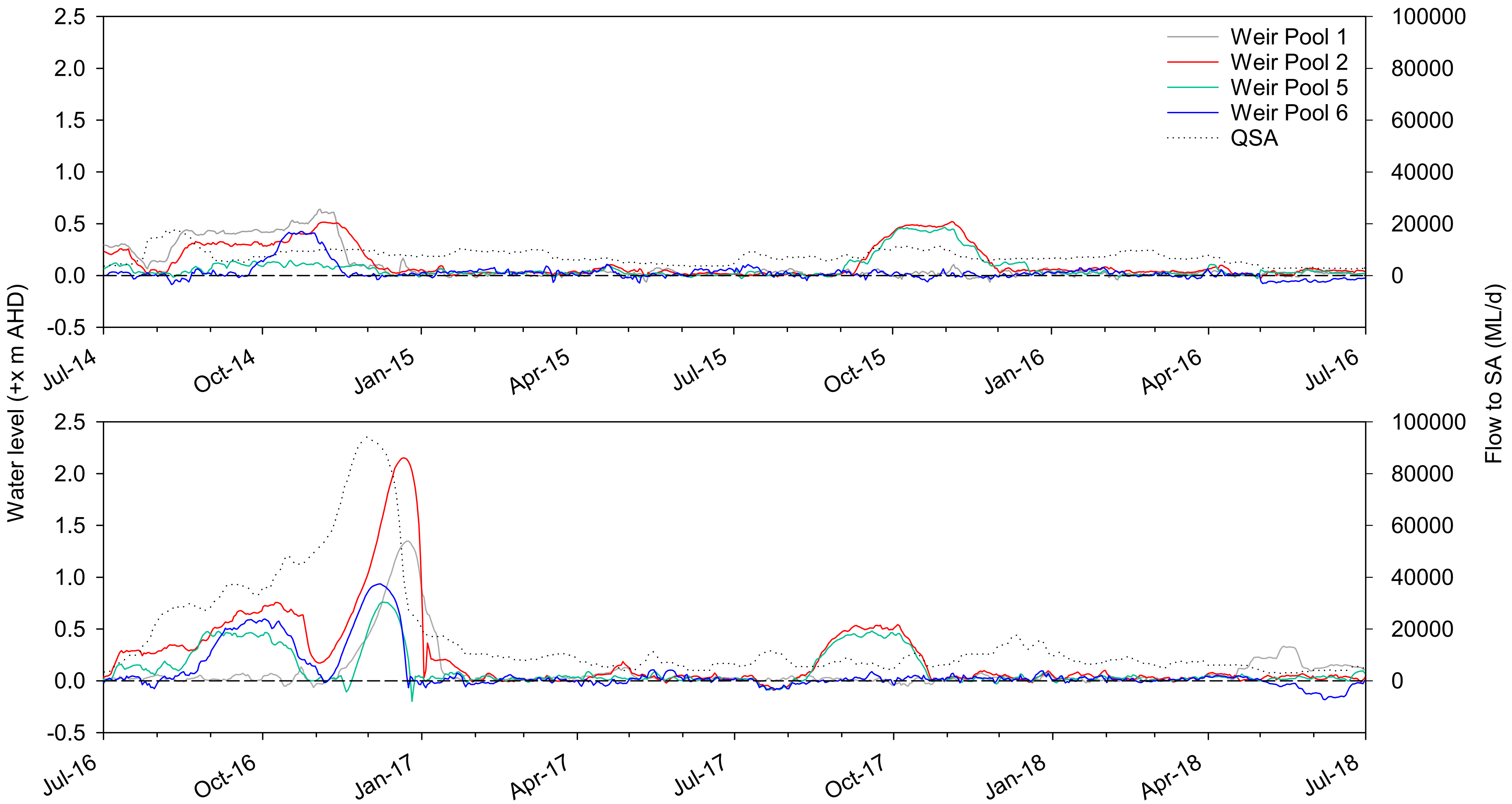

Figure B1. Water levels in the Lock 1, 2, 5 and 6 weir pools from 2014–2018, showing weir pool manipulations (DEW). Water levels are measured at Lock 1 US (A4260902) + 3.1 m AHD, Lock 2 US (A4260518) + 6.1 m AHD, Lock 5 US (A4260512) + 16.3 m AHD and Lock 6 US (A4260510) + 19.25 m AHD sites.

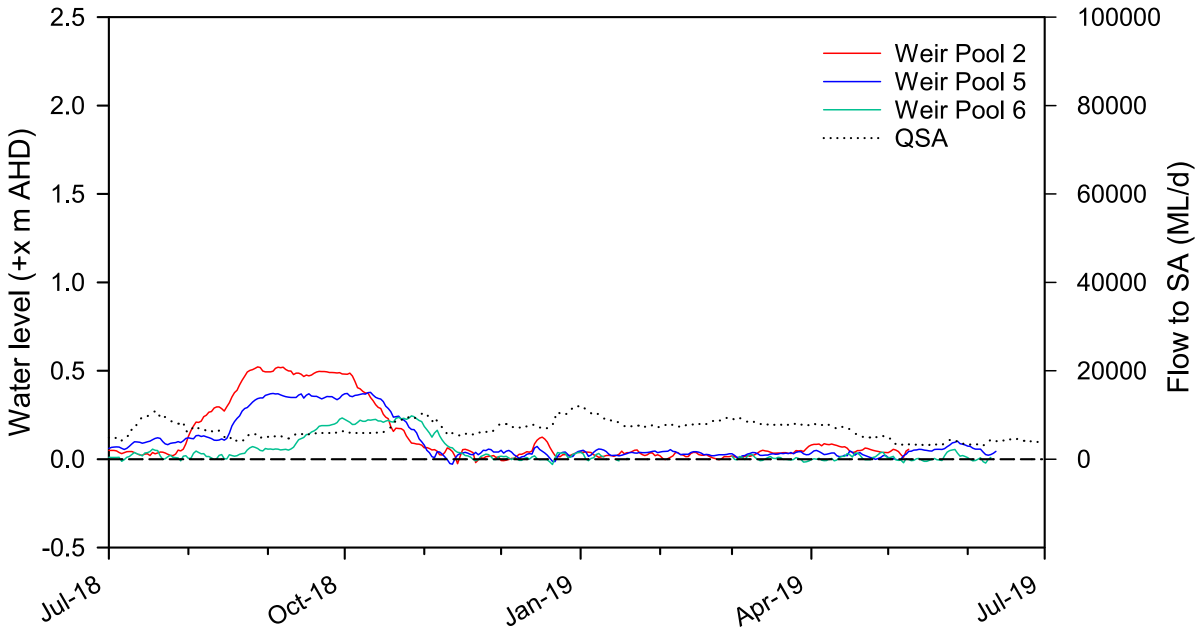

Figure B2. Water levels in the Lock 1, 2, 5 and 6 weir pools in 2018-19, showing weir pool manipulations (DEW). Water levels are measured at Lock 2 US (A4260518) + 6.1 m AHD, Lock 5 US (A4260512) + 16.3 m AHD and Lock 6 US (A4260510) + 19.25 m AHD sites.

Chowilla floodplain regulator operation

Regulated inundations of floodplains using the Chowilla regulator occurred in winter–spring 2014 and 2018, whilst in-channel rises in water level occurred in 2015 and 2018 (Table B2; Figure B3). The recession of these events commonly occurred in mid- to late spring, with water levels returning to normal level by early to mid-December. Whilst other environmental water (i.e. TLM and RMIF) or unregulated flow was used to undertake the regulator events, the concurrent passing of Commonwealth environmental water along the system during the Chowilla regulator events (e.g. 2014-15) supported maintenance of the required flow to South Australia. Refer to LTIM annual technical reports and http://www.environment.sa.gov.au for more detail.

Table B2. Chowilla floodplain regulator event details from 2014–2019 (Source: MDBA).

Year	Event	Peak water level (@ Chow reg.)	Timing	Supporting environmental water
2014-15	Inundation (2,300 ha)	19.1 m AHD	Peak in mid-October. Water recession occurred between early November and early December.	104 GL TLM
2015-16	In-channel rise	17.8 m AHD	Raising commenced early October, peaking in early November. From mid-November, stop logs in the Chowilla regulator were gradually removed to allow water levels to return to normal level by mid-December 2015.	3.2 GL TLM
2016-17	High-level testing	19.8 m AHD	Commencing in early August 2016, peak water levels 28 September and held at ~19.6 m AHD from mid-September to mid-October 2016. Water level recession began shortly after peak levels in mid-October 2016.	O GL (Unregulated flows fully supported the event)
2017-18	None			N/A
2018-19	In-channel rise	18.6 m AHD	Commencing in mid-August 2018, with peak water levels between early and late October. Water level recession commenced afterwards and returned to normal levels by early December 2018.	35.57 GL RMIF

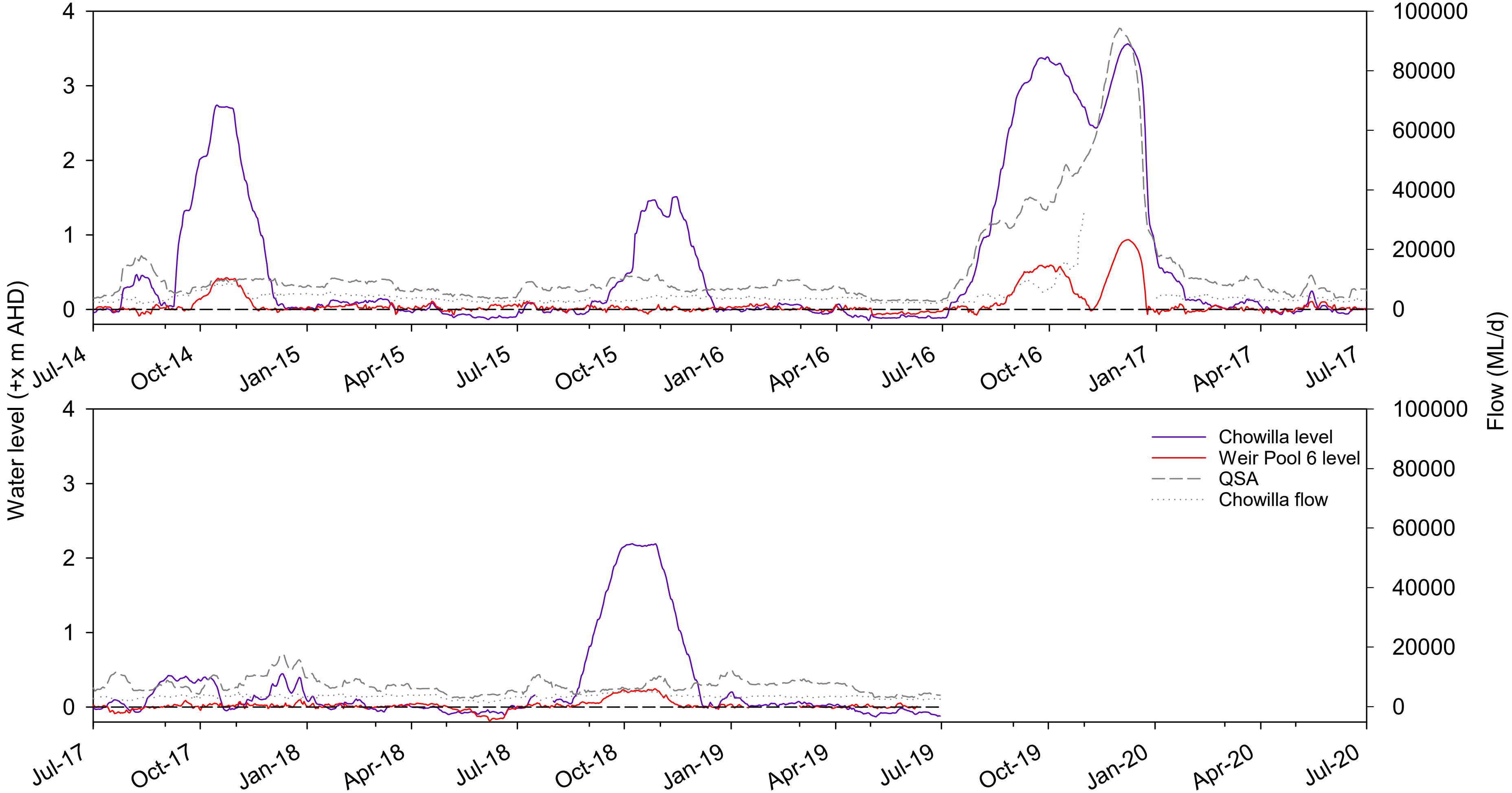

Figure B3. Water levels in Chowilla Creek, downstream of Monomon Creek (+16.4 m AHD), and upstream of Lock 6 (+19.25 m AHD) from 2014–2019, showing Chowilla regulator operations (DEW). Flow to South Australia (QSA) and within Chowilla Creek (calculated as QSA minus flow downstream of Lock 6) are presented.

Watering and management activities outside of the LMR

Manipulation of water levels in weir pools above the LMR

Water levels in Weir Pools 7, 8, 9 and 15 have been raised and/or lowered annually from 2014–2019, except for 2014-15 when only Weir Pools 8 and 9 were raised and lowered (Table B3). During 2014-15, 2015-16, 2017-18 and 2018-19, Commonwealth environmental water was used to account for 'net' use, i.e. combined loss from raising and savings from lowering for the duration of the environmental watering event and for all weirs involved in the event. In 2016-17, weir pools were re-filled by unregulated flow following weir pool manipulations. However, it was Commonwealth environmental water underwriting the requirement of the weir-pools re-filling that enabled the river operators to undertake the manipulations.

Refer to LTIM annual technical reports for more detail regarding the timing and magnitude of the events.

Table B3. Weir pool manipulation timing and magnitude details above the LMR from 2014–2019. NPL = normal pool level, FSL = full supply level. *indicates years when Commonwealth environmental water accounted for losses.

Year	Weir pool			
	7	8	9	15
2014-15*		Raising of weir pool by up to +0.8 m NPL (August to November 2014). Lowering of weir pool by up to -0.8 m NPL (January to April 2015).	Raising of weir pool by up to +0.2 m NPL (August to November 2014). Lowering of weir pool by up to -0.1 m NPL (January to March 2015).	
2015-16*	Raising of weir pool to +0.6 m NPL (August 2015 to January 2016). Lowering of weir pool to -0.8 m NPL (January to May 2016).	Raising of weir pool to +0.8 m NPL (August 2015 to mid-December 2015). Lowering of weir pool to -0.8 m NPL (December 2015 to May 2016).	Raising of weir pool to +0.25 m NPL (July to September 2015). Lowering of weir pool to -0.1 m NPL (brief period of -0.2 m NPL) (October 2015 to February 2016).	Raising of weir pool to +0.6 m NPL (July to December 2015). Lowering of weir pool to -0.3m NPL (April to June 2016).
2016-17	Operational range: +0.55 m above to -0.9m below NPL.	Operational range: +0.85 m above to -1.0 m below NPL.	Operational range: +0.24 m above to -0.1 m below NPL.	Operational range: +0.6 m above to -0.45 m below NPL.
2017-18*	Raising of weir pool up to +0.55 m above FSL (spring raising).	Raising of weir pool up to +0.35 m above FSL (spring raising).	Raising of weir pool up to +0.23 m above FSL (spring raising).	Raising of weir pool to +0.35 m above FSL

Year	Weir pool			
	7	8	9	15
	Lowering of weir pool up to -0.55 m below FSL (winter lowering), and up to -0.10m below FSL (autumn lowering).	Lowering of weir pool up to -1.00 m below FSL (winter lowering), and up to -0.32m below FSL (autumn lowering).	Lowering of weir pool up to -0.12 m below FSL (autumn lowering).	(spring-summer raising). Lowering of weir pool to -0.45 m below FSL (winter lowering).
2018-19*	Raising of weir pool up to +0.3 m above FSL (spring raising). Lowering of weir pool up to -0.5 m below FSL (summer lowering), and up to -0.9m below FSL (autumn lowering).	Lowering of weir pool up to -0.3 m below FSL (winter lowering), up to -0.5 m below FSL (spring lowering), up to -0.6 m below FSL (summer lowering), up to -1.0 m below FSL (autumn lowering).	Lowering of weir pool up to -0.1 m below FSL (late autumn and winter lowering).	Lowering of weir pool to -0.3 m below FSL (winter lowering).

In 2016/17, water levels in Weir Pools 7, 8, 9 and 15 were manipulated (raised and/or lowered), within operational limits, to introduce a more natural wetting and drying cycle for the benefit of the riverine environment by increasing variability in river levels. No environmental water was used to raise weir pools above NPL because the initial managed raisings were achieved by unregulated flows.

Other watering events and management actions

From 2014–2019, environmental water was delivered to the LMR channel, primarily as deliveries from return flows through coordinated watering events across the southern connected Basin, to achieve multi-site environmental outcomes. The major upstream watering events that were supported by environmental water and may be relevant to the evaluation in this report are summarised in Table B4. Refer to LTIM annual technical reports for more detail.

Table B4. Details for upstream watering events and management actions supported by environmental water (eWater) from 2014–2019. CEW = Commonwealth environmental water, TLM = The Living Murray, VEWH = Victorian Environmental Water Holder, IVT = Inter-Valley Transfer, NSW DPIE = New South Wales Department of Planning, Industry and Environment, RMIF = River Murray Increased Flows.

Year	Event	Event description and timing	Supporting eWater
2014-15	Hume releases	From 22–30 June 2015, environmental water was delivered from Hume Dam to contribute to in-channel flows in the Murray River main channel.	CEW
	Goulburn River flows	Environmental water was delivered to the lower Goulburn River channel during 2014/15 to produce winter base flows/freshes, and spring and autumn freshes.	CEW, IVT, TLM, VEWH
2015-16	B-M Forest inundation	From July–September 2015, overbank flows (with two small freshes) occurred in the Barmah and Millewa Forests. From 11 September to 31 October 2015, environmental water was provided to maintain flows and extend the inundation of Millewa Forest (source, CEWO). A gradual recession followed shortly after.	TLM and CEW
	Goulburn River flow pulse	Environmental water was delivered to the lower Goulburn River channel during 2015/16 to produce a spring pulse and autumn fresh.	CEW, TLM, VEWH and IVT
2016-17	Rufus River refuge habitat	Releases were made from Lake Victoria, during November and December, to provide refuge habitat for aquatic fauna downstream of the Darling junction. From 17 to 31 December, Lake Victoria outflow into the Rufus River was supplemented by environmental water.	CEW and TLM
	Yarrawonga flow pulse	Environmental water was delivered from Hume in November and December to the Murray River main channel. From January onwards, environmental water maintained water levels in the River to enable water flow into the creeks of Barmah–Millewa. From 12 February to 5 March 2017, CEW contributed to a fish pulse (followed by pulses in the Goulburn and Campaspe Rivers), targeting downstream of Yarrawonga.	CEW and TLM

Year	Event	Event description and timing	Supporting eWater
2016-17	Goulburn River fresh	Environmental water contributed towards an autumn fresh in the Goulburn River between late February and early April 2017.	CEW, TLM, VEWH and IVT
	Darling River flow	CEW was delivered from early December 2016 to early January 2017, and from late April to late June 2017, while TLM water was delivered from mid-September to late November 2016 Flows peaked at Weir 32 in early January during operational releases. Also in 2016/17, CEW and environmental water from the NSW DPIE were delivered down the Great Darling Anabranch between February and June 2017.	CEW, TLM and NSW DPIE
2017-18	Barmah-Millewa Forest inundation	Between August and December 2017, environmental water was delivered from Hume Dam. Concurrently, from August to October 2018, environmental water was delivered through Barmah–Millewa Forest regulators (source, CEWO). During October and November, environmental water contributed to overbank flows through the Barmah Millewa Forest, with flows that returned to the river being delivered to South Australia.	CEW, LTM, RMIF and NSW DPIE
	Victorian tributary return flows (e.g. Goulburn River)	Environmental water was delivered to the Goulburn River during 2017-18 (source, CEWO), promoting a July winter fresh and two spring freshes in early October and late November. Flow in the Goulburn River peaked in early December as a result of an unregulated flow event. Following this event, a small portion of the environmental water was delivered on the flow recession for blackwater mitigation. From February to early June 2018, IVT accounted for the majority of flows in the Goulburn River, including a winter fresh commencing in late June 2018 and extending into August 2018. Return flows from other Victorian tributaries also contributed to environmental flow to South Australia during 2017-18 (e.g. Campaspe River).	CEW, TLM, VEWH and IVT
	Murrumbidgee River flows	During July and August 2017, environmental water was delivered to reconnect the Murrumbidgee River with mid-Murrumbidgee wetlands, areas of the Lowbidgee floodplain and the Murray Junction Wetlands. Return flows reached South Australia during August and September 2017.	CEW and NSW DPIE
	Lower Darling River flows	In 2017-18, environmental water was delivered to the Lower Darling River, providing base flows for fish habitat in early spring (TLM) and assisting in shaping the recession of operational releases in late spring–early summer (TLM and CEW). Return flows reached South Australia during July to October 2018.	CEW and TLM

Year	Event	Event description and timing	Supporting eWater
2017-18	Lower Lakes drawdown	A managed ‘drawdown’ of Lower Lakes water levels occurred in autumn 2018. CEW was used to protect against water levels dropping below 0.5m.	CEW
2018-19	Barmah-Millewa Forest inundation	Water delivery through the Barmah-Millewa Forest regulators commenced in mid-July. RMIF covered use (losses) in the regulators throughout July and August. CEW was used to underwrite overbank ‘initial’ losses in the Barmah-Millewa Forest when regulators were left open from 2–14 September 2018.	RMIF and CEW
	Hume releases	Environmental water was delivered from Hume Dam throughout 2018-19 to provide in-stream variable releases to the Murray River main channel.	CEW, TLM, RMIF, NSW DPIE

APPENDIX C: DEW EVALUATION QUESTIONS

Table C1. DEW short-term (one-year) and long-term (five-year) evaluation questions for CEWO LTIM indicators. Evaluation questions are based on ecological targets from the Long-Term Environmental Watering Plan (LTWP) for the South Australian Murray River. DEW evaluation questions serve as 'additional' questions as there may be some CEWO questions that are also relevant to DEW's targets from the LTWP. CEW = Commonwealth environmental water; eWater = environmental water. See annual CEWO LTIM LMR evaluation reports for answers to short-term questions for 2015-16–2017-18.

Contribution (to what extent CEW contributed towards the outcome, with the significance of the outcome considered):

⬜ Unknown 🟥 Negative ⬜ None/negligible 🟩 Minor 🟩 Moderate 🟦 Substantial

Hydrology (Channel) and Hydrological Regime (modelling)

<table>
<tr><th rowspan="2">DEW evaluation questions</th><th colspan="5">Outcomes of CEW delivery (2014-15–2018-19)</th></tr>
<tr><th>2014-15</th><th>2015-16</th><th>2016-17</th><th>2017-18</th><th>2018-19</th></tr>
<tr><td rowspan="2">What did CEW contribute to providing a seasonal hydrograph that encompassed variation in discharge, velocity and water levels?</td><td>🟩</td><td>🟩</td><td>🟩</td><td>🟩</td><td>🟩</td></tr>
<tr><td colspan="5">This evaluation question has been answered by considering the combination of the questions "What did CEW contribute to hydraulic diversity within weir pools?" and "What did CEW contribute to variability in water levels within weir pools?" in Table 4.</td></tr>
<tr><td rowspan="2">What did CEW contribute to meeting the EWRs (all metrics) for the Channel?</td><td>0/0</td><td>1/0</td><td>9/9</td><td>1/0</td><td>0 EWR Met with CEW/0 EWR Met without CEW</td></tr>
<tr><td colspan="5">The 10,000 ML/d for 60 day Environmental Watering Requirement (EWR) was met in 2 of the 5 years. Without CEW contributions, this EWR would not have been met in these years. In the 2016-17 high flow year EWRs were met by the unregulated flow.
Total number of EWRs is 7 for the channel + 5 floodplain = 12 (DEWNR 2015).</td></tr>
</table>

DEW evaluation questions	Outcomes of CEW delivery (2014-15–2018-19)				
	2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to providing diverse hydraulic conditions and complex habitat for flow dependant biota and processes?					
	This evaluation question has been considered equivalent to the "What did CEW contribute to hydraulic diversity within weir pools?" evaluation question in Table 4.				
What did CEW contribute to providing diverse hydraulic conditions over the range of velocity classes in the lower third of weir pools so that habitat and processes for dispersal of organic and inorganic material between reaches are maintained?					
	Analysis of velocity results and expert elicitation has identified that this evaluation question is met by the 20,000 ML/d for 60 days EWR. This EWR was only met in 2016/17, and in that high flow year the EWR would have been met without CEW contributions.				
What did CEW contribute to the range of velocity classes being present in the lower third of weir pools for at least 60-days in spring/summer in at least 3 years out of 5?	This evaluation question adds requirements around timing, duration and frequency to the above. In line with the above question, CEW made a negligible contribution to this question.				

Stream Metabolism

DEW evaluation questions	Outcomes of CEW delivery (2014-15–2018-19)				
	2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to temporarily shifting open water productivity towards heterotrophy? What did CEW contribute to increased nutrients and DOC levels?	This could not be assessed as the water source and quality of modelled flows without CEW contributions are unknown. In particular, the differences in DOC concentrations attributable to CEW could not be assessed. However, efforts are underway to explore the changes in DOC associated with enhanced lateral connectivity and it may be possible to draw conclusions from this regarding CEW contributions to connectivity and the influence on heterotrophy.				
What did CEW contribute to maintaining or increasing annual autotrophic production and increasing annual heterotrophy?	1.01	1.02	1.02	1.02	1.02
	Increased flows generally reduced the volumetric rate of primary production, but increased the cross-sectional rates which increased the overall "carrying capacity" of the river. At the LMR sites, the listed fractional increases in cross-sectional GPP due to environmental flows were negligible due to the fixed water levels set by weirs. An unregulated site showed substantial changes, but was modelled and not measured.				
What did CEW contribute to maintaining dissolved oxygen levels above 50% saturation throughout the water column at all times?	0	53	21	50	25
	Environmental flows decreased the likelihood of low DO by increasing water mixing and oxygen exchange at the surface. This was assessed as the extra days per year with water velocities > 0.18 m/s due to environmental flows. A substantial contribution was considered greater than 30 days, moderate 15-30 days, minor 7-14 days and negligible < 7days.				

Matter Transport (modelling)

DEW evaluation questions	Outcomes of CEW delivery (2014-15–2018-19)				
	2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to providing for the dispersal of organic and inorganic material and organisms between river and wetlands?	591,54 and 6.4	1038, 95 and 6.7	122, 11 and 14	1513, 140, 13	CEW contributed to the export of 872, 85 and 4.2 tonnes of particulate organic nitrogen, particulate organic phosphorus and phytoplankton, respectively, over the barrages.
	The values above are respectively tonnes of Total Organic nitrogen, Total Organic Phosphorus and Cha exported over the barrages due to CEW water. The modelling suggests that CEW increased the export of dissolved and particulate matter. This was observed through: • Increased exports of nutrients from the Murray River Channel, Lower Lakes and Coorong/Murray Mouth. • Increased exports of phytoplankton biomass from the Murray River Channel, Lower Lakes and Coorong/Murray Mouth. It is important to remember than nutrients are a resource that drive productivity and fuel food webs. The increased transport of dissolved and particulate matter may have provided benefits for the Lower Lakes, Coorong and near-shore marine environment by providing energy to ecosystem productivity. Nutrients are incorporated in phytoplankton that are consumed by higher trophic organisms.				

<table>
<tr><th>DEW evaluation questions</th><th>Outcomes of CEW delivery (2014-15–2018-19)</th></tr>
<tr><td>What did CEW contribute to the salinity regime, including maintaining the average electrical conductivity in Lake Alexandrina below 700 (0.45), 1000 (0.64) and 1500 (0.96) µS/cm (PSU*)?

*In fresh conditions, PSU is essentially equivalent to ppt and 1 µS/cm is approximately 640 ppm.</td><td>The modelling suggests that CEW impacted positively on the concentrations of dissolved and particulate matter. This was observed through a reduction in salinity in Lake Alexandrina. The values below are modelled median PSU in the middle of Lake Alexandrina.
<table>
<tr><th></th><th>2014-15</th><th>2015-16</th><th>2016-17</th><th>2017-18</th><th>2018-19</th></tr>
<tr><td>With all water</td><td>0.29</td><td>0.29</td><td>0.24</td><td>0.31</td><td>0.33</td></tr>
<tr><td>No CEW</td><td>0.33</td><td>0.39</td><td>024</td><td>0.35</td><td>0.45</td></tr>
<tr><td>No eWater</td><td>0.35</td><td>0.43</td><td>0.24</td><td>0.36</td><td>0.50</td></tr>
</table>
The CEW water helped prevent a breach in achieving salinity targets in 2018-19 where there was a modelled median salinity of 0.33 practical salinity units (PSU) with all water, compared to 0.45 PSU without CEW.</td></tr>
<tr><td>What did CEW contribute to establishing and maintaining stable salinities in the lakes and a variable salinity regime in the Murray estuary and Coorong?</td><td>Riverine flows that exit the barrages play a role is exporting salt from the basin. However, they also play a role in reducing the amount of salt that enters through the Murray Mouth. When flow is high, there is a net export of salt from the Murray Mouth, but when flow is low, salt can intrude into the Murray Mouth.

In 2014-15, 2015-16, 2017-18 and 2018-19, there was a net import of salt through the Murray Mouth and into the Coorong, which increased the salinity of the Coorong. CEW significantly reduced salt import into the Coorong. Without CEW, there would have been extreme salinity in the South Lagoon of the Coorong and loss of key species.</td></tr>
<tr><td>What did CEW contribute to ensuring adequate flushing of salt from the Murray to the Southern Ocean as measured by the Basin Plan target of >2 million tonnes of salt export for a three-year rolling average?</td><td>Flow has been relatively low in four of the five years of LTIM monitoring. In the low flow years (2014-15, 2015-16, 2017-18 and 2018-19), CEW played a key role in salt export from the Basin, accounting for 64, 87, 69 and 70% of salt export, respectively. In the low flow years, the salt export ranged 228,293–446,855 tonnes, which was well below the Basin Plan target of 2 million tonnes of salt.

In the high flow year (2016-17), 1.5 million tonnes was exported and CEW contributed 8% of salt export.</td></tr>
</table>

Micro-invertebrates

<table>
<tr><th rowspan="2">DEW evaluation questions</th><th colspan="4">Outcomes of CEW delivery (2014-15–2018-19)</th></tr>
<tr><th>2014-15</th><th>2015-16</th><th>2016-17</th><th>2017-18</th></tr>
<tr><td rowspan="2">What did CEW contribute to increased microinvertebrate input from floodplain to the river and thus reducing the reliance of in-stream food webs on autochthonous productivity?</td><td>71%</td><td>58%</td><td>68%</td><td>56%</td></tr>
<tr><td colspan="4">Of taxa recorded from the LMR main channel during in-channel flows in 2014-15, 2015-16 and 2017-18, 71, 58 and 56%, respectively, were not true potamoplankton (plankton of flowing waters), but littoral/epiphytic/epibenthic incursions, flushed into the main channel from floodplain or littoral sources. In contrast, 68% taxa were not true potamoplankton during sampling in 2016-17, when there were overbank flows. CEW return flows, and flooded littoral/riparian margins (e.g. Chowilla Anabranch and Weir Pools 5, 2, 7, 8 and 9) likely translocated some of these taxa, some in appreciable numbers, into the main channel. CEW likely had less of a role in translocating taxa in 2016-17, during high, unregulated, overbank flows.</td></tr>
<tr><td rowspan="2">What did CEW contribute to increased dispersal of organisms between river and wetlands?</td><td>Nil</td><td>Nil</td><td>Nil</td><td>Nil</td></tr>
<tr><td colspan="4">No wetland samples were collected in 2014-15, 2015-16, 2016-17 or 2017-18 to ascertain CEW dispersal of microinvertebrates from the main channel flows.</td></tr>
</table>

Fish Spawning and Recruitment

DEW evaluation questions	Outcomes of CEW delivery (2014-15–2018-19)				
	2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
What did CEW contribute to the population age structure of golden perch in the LMR?	Nil	Nil	Nil	Nil	CEW delivery in 2018-19 did not contribute to the presence of any new cohorts (age 0+) of golden perch in the LMR, despite spawning during spring–early summer 2018.
Did CEW contribute to the population age structure of golden perch in the LMR Selected Area so that YOY, sub-adults and adults were present in 3 of the last 4 years?	No. CEW did not contribute to the presence of any new cohorts of golden perch from 2014–2019. YOY (age 0+) golden perch were not sampled in 2014-15, 2015-16, 2017-18 or 2018-19. YOY golden perch sampled by fyke netting in 2016-17 had birth dates coinciding with the period of high unregulated flows.				
What did CEW contribute to the population age structure of silver perch in the LMR?	Nil	Nil	Nil	Nil	CEW delivery in 2018-19 did not contribute to the presence of any new cohorts (age 0+) of silver perch in the LMR. No silver perch spawning was detected in 2018-19.
Did CEW contribute to the population age structure of silver perch in the LMR Selected Area so that YOY (age 0+), sub-adults and adults were present in 3 of the last 4 years?	No. CEW did not contribute to the presence of any new cohorts of silver perch from 2014–2019. YOY (age 0+) were not sampled in 2014-15, 2015-16, 2016-17, 2017-18 or 2018-19.				
Did CEW contribute to a YOY or age 1+ cohort that represented >30% of the golden perch population in the LMR?	No	No	No	No	No. Age 0+ (2018-19) and 1+ (2017-18) cohorts represented <30% of the golden perch population in the LMR during autumn 2019. In 2018-19, there was spawning of golden perch, but negligible recruitment.

DEW evaluation questions	Outcomes of CEW delivery (2014-15–2018-19)				
	2014-15	**2015-16**	**2016-17**	**2017-18**	**2018-19**
Did CEW contribute to at least 2 large golden perch recruitment events in the last 4 years, demonstrated by a cohort representing >30% of the population from the LMR Selected Area?	No. Two large golden perch recruitment events in the last 4 years were not observed in the LMR Selected Area. The LMR population from 2015–2019 was dominated by three cohorts spawned during 2009-10, 2010-11 and 2011-12.				
Did CEW contribute to a YOY or age 1+ cohort that represented >30% of the silver perch population in the LMR?	No	No	No	No	Age 0+ (2018-19) and 1+ (2017-18) cohorts represented <30% of the silver perch population in the LMR during autumn 2019. A larger sample size will provide a more reliable indication of the relative abundance of YOY and age 1+ silver perch in the LMR.
Did CEW contribute to at least 2 large silver perch recruitment events in the last 4 years, demonstrated by a cohort representing >30% of the population from the LMR Selected Area?	No. Two large silver perch recruitment events in the last 4 years were not observed in the LMR Selected Area. A larger sample size will provide a more reliable indication of the relative abundance of YOY and age 1+ silver perch in the LMR.				

Fish Assemblage: Fish (Channel) data have been consolidated to evaluate a number of fish targets of DEW's LTWP. These questions and answers do not relate to evaluation of flow or CEW. Furthermore, the LTIM Fish monitoring program is not designed to determine what is facilitating changes in population dynamics of fish species for DEW's LTWP evaluation questions, e.g. spawning and recruitment of Murray cod or common carp.

Outcome: 🟥 = negative; 🟩 = positive; ▨ = unable to be detected.

DEW evaluation questions	**Answers to evaluation questions**				
	2015	**2016**	**2017**	**2018**	**2019**
Did the length-frequency distribution for Murray cod in the Gorge zone reflect recent recruits, sub-adults and adults?	X	X	X	X	No. During autumn 2019, recent recruits (i.e. <300 mm TL) were sampled in the Gorge geomorphic zone of the LMR; however, sub-adults (i.e. 300–600 mm TL) and adults (>600 mm TL) were not sampled.
Did the length-frequency distribution for Murray cod in the Gorge zone reflect recent recruits, sub-adults and adults during 4 of the last 5 years?	No. The length-frequency distribution for Murray cod in the Gorge zone did not reflect recent recruits, sub-adults and adults during any one year. Additional targeted sampling is likely required to adequately sample sub-adult and adults.				
Did a YOY cohort represent >50% of the Murray cod population from the Gorge zone?	√	√	√	√	Yes. During autumn 2019, only YOY (i.e. <150 mm TL) Murray cod were sampled, which represented 100% of the population in the Gorge geomorphic zone of the LMR.
Did the length-frequency distribution for Murray cod indicate at least 1 large recruitment event in the last 5 years, demonstrated by a YOY cohort representing >50% of the population from the Gorge zone?	Yes. The length-frequency distribution for Murray cod indicated a large recruitment event in all 5 years, demonstrated by a YOY cohort representing >50% of the population from the Gorge zone?				

DEW evaluation questions	**Answers to evaluation questions**				
	2015	**2016**	**2017**	**2018**	**2019**
Did the abundance of Murray cod in the Gorge zone increase by ≥20% over a 5-year period?	Yes. The abundance of Murray cod in the Gorge zone increased by 89% over the 5-year period from 0.033 individuals/90 s (± 0.007 S.E.) in 2015 to 0.063 (± 0.013) in 2019.				
Did the abundance of golden perch in the Gorge zone increase by >30% over a 5-year period?	No. The abundance of golden perch in the Gorge zone decreased by 45% over the 5-year period from 0.566 individuals/90 s (± 0.076 S.E.) in 2015 to 0.313 (± 0.021) in 2019.				
Did the abundance of silver perch in the Gorge zone increase by >30% over a 5-year period?	No. The abundance of silver perch in the Gorge zone decreased by 42% over the 5-year period from 0.012 individuals/90 s (± 0.007 S.E.) in 2015 to 0.007 (± 0.007) in 2019. These results should be interpreted with caution, given the large error estimate.				
Did the abundance of freshwater catfish in the Gorge zone increase by ≥30% over a 5-year period?	No. The abundance of freshwater catfish in the Gorge zone decreased by 23% over the 5-year period from 0.018 individuals/90 s (± 0.009 S.E.) in 2015 to 0.014 (± 0.008) in 2019. These results should be interpreted with caution, given the large error estimate.				
Did the length-frequency distribution for bony herring include size classes representing YOY in the Gorge zone?	N/A	√	√	√	Yes. During autumn 2019, length-frequency distributions indicated YOY were present for bony herring.
Did the length-frequency distribution for bony herring in the Gorge zone include size classes representing YOY during all 4* years?	Yes. The length-frequency distribution for bony herring in the Gorge zone include size classes representing YOY during all 4 years.				
Did the length-frequency distribution for Murray rainbowfish and carp gudgeon, include size classes representing YOY in the Gorge zone?	√	√	√	√	Yes. During autumn 2019, length-frequency distributions indicated YOY were present for Murray rainbowfish and carp gudgeon.

DEW evaluation questions	Answers to evaluation questions				
	2015	2016	2017	2018	2019
Did the length-frequency distribution for Murray rainbowfish and carp gudgeon in the Gorge zone include size classes representing YOY during all 5 years?	Yes. The length-frequency distribution for Murray rainbowfish and carp gudgeon in the Gorge zone include size classes representing YOY during all 5 years.				
Did the relative abundance of common carp in the Gorge zone increase during the current year, relative to the previous year, whilst the relative abundances of flow-dependent native species decreased?**	N/A	√	√	X	No. There was a decrease in the ratio (total abundance) of common carp to flow-dependant, native species (golden perch and silver perch) at nine of the ten sites sampled in 2019, relative to the previous year. During 2018 the mean site ratio was 3.64 carp (± 0.94 S.E.) to every 1 flow-dependant, native species. In 2019, this ratio decreased to 1.69 carp (± 0.22) to every 1 flow-dependant, native species.
Did the relative abundance of common carp in the Gorge zone increase over a 5-year period, whilst the relative abundances of flow-dependent native species decreased?**	Yes. There was an increase in the ratio (total abundance) of common carp to flow-dependant, native species (golden perch and silver perch) at nine of the ten sites sampled in 2019, relative to 2015. This was associated with strong recruitment of common carp in 2016-17 and a small but steady decrease in golden perch abundance over the five-year period. During 2015 the mean site ratio was 0.58 carp (± 0.09 S.E.) to every 1 flow-dependant, native species. In 2019, this ratio increased to 1.69 carp (± 0.22) to every 1 flow-dependant, native species.				
Did the estimated biomass of common carp in the Gorge zone increase during the current year, relative to the previous year, whilst the estimated biomass of flow-dependent native species decreased?**	N/A	√	√	√	No. There was a decrease in the ratio (total biomass) of common carp to flow-dependant, native species (golden perch and silver perch) at eight of the ten sites sampled in 2019, relative to the previous year. During 2018, the mean site ratio was 3.14 kg of carp (± 1.07 S.E.) to every 1 kg of flow-dependant, native species. In 2019, this ratio decreased to 1.52 kg of carp (± 0.23) to every 1 kg of flow-dependant, native species.

DEW evaluation questions	**Answers to evaluation questions**				
	2015	**2016**	**2017**	**2018**	**2019**
Did the estimated biomass of common carp in the Gorge zone increase over a 5-year period, whilst the estimated biomass of flow-dependent native species decreased?**	Yes. There was an increase in the ratio (total biomass) of common carp to flow-dependant, native species (golden perch and silver perch) at seven of the ten sites sampled in 2019, relative to 2015. During 2015 the mean site ratio was 1.32 kg of carp (± 0.32 S.E.) to every 1 kg of flow-dependant, native species. In 2019, this ratio increased to 1.52 kg of carp (± 0.23) to every 1 kg of flow-dependant, native species.				

*Bony herring were not assessed as a target species during 2014.

** To remove sampling season bias, only sites sampled during autumn 2017 were used in carp ratio comparisons against 2018. Site ratios of common carp to flow-dependant, native species were calculated by dividing the total biomass or number of individuals (abundance) of carp for that site by the total biomass or number of individuals (abundance) of golden perch and silver perch for the same site, respectively. The mean site ratio for a particular year was calculated by averaging the site ratios. Common carp were not weighed as part of the Fish (channel) sampling, so biomass was estimated by converting fork lengths to weights based on a FL–mass equation in Vilizzi and Walker (1999).

APPENDIX D: SUPPLEMENTARY INFORMATION FOR INDICATORS

Table D1. Median concentrations and loads (tonnes) of salinity, nutrients and chlorophyll *a* during 2018-19 for the modelled scenarios at three selected sites. Scenarios include with all water, without Commonwealth environmental water (no CEW) and without any environmental water (no eWater).

Median concentrations

Site	**Scenario**	**Salinity (PSU)**	**Ammonium (mg/L)**	**Phosphate (mg/L)**	**Silica (mg/L)**	**Particulate organic nitrogen (mg/L)**	**Particulate organic phosphorus (mg/L)**	**Chlorophyll *a* (ug/L)**
Wellington	With all water	0.1677	0.0011	0.0014	2.2496	0.8529	0.0645	13.0585
	No CEW	0.1880	0.0012	0.0016	3.5793	0.9540	0.0770	12.4740
	No eWater	0.1962	0.0012	0.0016	4.8930	1.0463	0.0899	11.9751
Lake Alexandrina Middle	With all water	0.3312	0.0094	0.0003	15.0916	1.9508	0.1939	9.0184
	No CEW	0.4459	0.0095	0.0004	24.8080	2.7847	0.2820	10.8069
	No eWater	0.5027	0.0012	0.0009	31.2121	3.2783	0.3360	11.9248
Murray Mouth	With all water	30.8970	0.0203	0.0017	3.4869	1.1426	0.0851	11.3620
	No CEW	34.8363	0.0192	0.0020	1.8060	1.0379	0.0714	11.9905
	No eWater	35.3037	0.0197	0.0023	1.3040	1.0110	0.0682	11.9977

Net load (tonnes)

Site	**Scenario**	**Salt**	**Ammonium**	**Phosphate**	**Silica**	**Particulate organic nitrogen**	**Particulate organic phosphorus**	**Chlorophyll *a***
Wellington	With all water	228,675.7243	0.5862	4.9095	2,991.4199	1,203.3013	102.1509	20.4404
	No CEW	162,625.3420	0.5123	3.1982	2,439.0615	827.5786	72.4672	12.8486
	No eWater	132,926.7341	0.3873	2.6802	2,180.1822	689.7025	60.8309	10.1043
Barrage	With all water	228,293.1906	12.0159	0.3447	10,582.6127	1,291.3049	128.1479	5.8175
	No CEW	67,396.4806	1.9834	0.1134	3,819.1542	419.1614	42.6696	1.6155
	No eWater	0	0	0	0	0	0	0
Murray Mouth	With all water	-2,680,573.8890	-7.5052	-2.1753	10,334.6556	975.6278	101.1605	5.6611
	No CEW	-5,151,627.3600	-18.4946	-2.6888	4,865.3458	159.2676	23.6880	1.8076
	No eWater	-5,438,074.6550	-21.8588	-3.0736	1,630.3360	-242.2370	-16.5038	0.5197

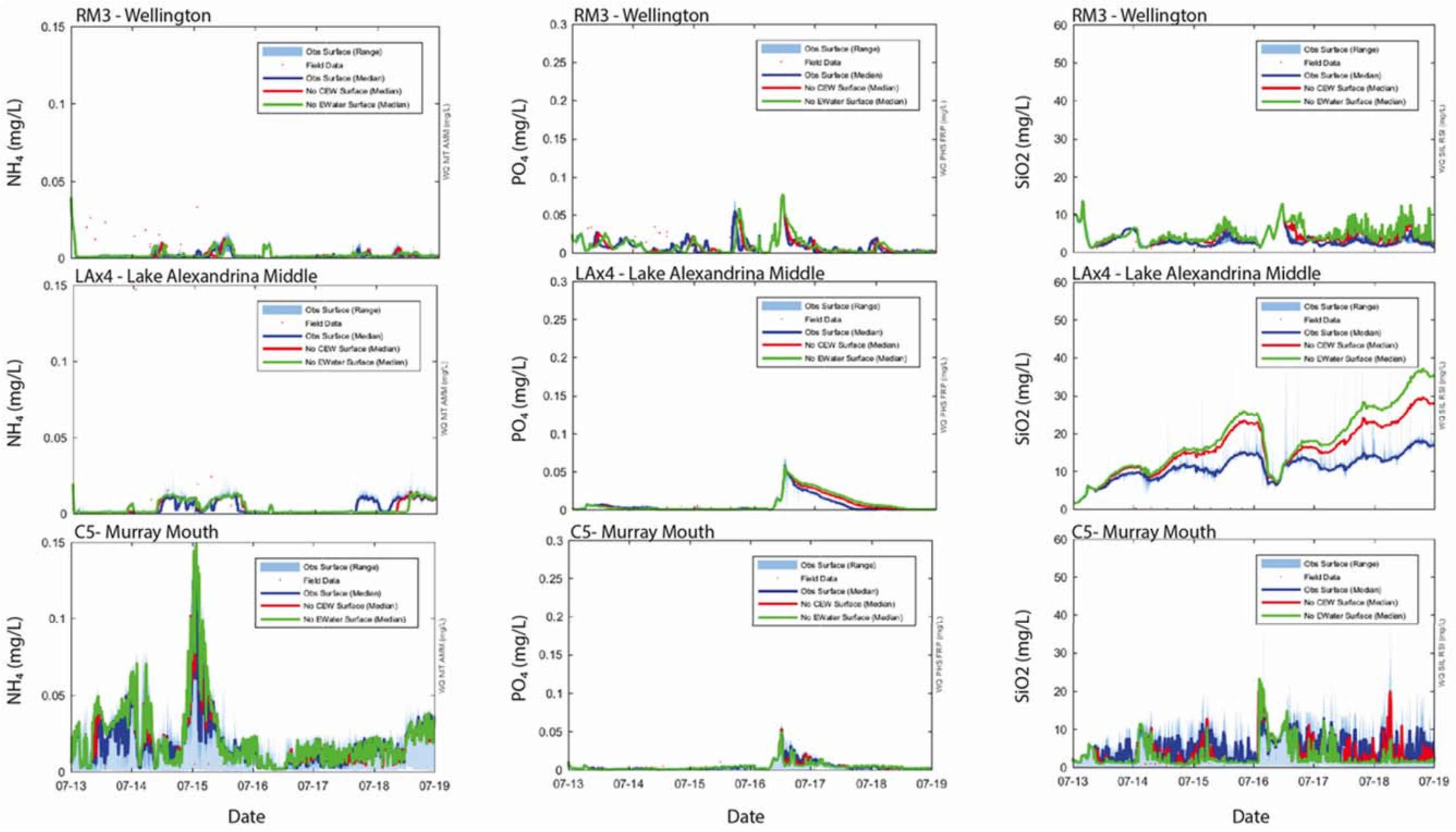

Figure D1. Daily concentrations of dissolved nutrient forms (ammonium, phosphate and silica) at Wellington, Lake Alexandrina and the Murray Mouth under three different flow scenarios from 2013–2019.

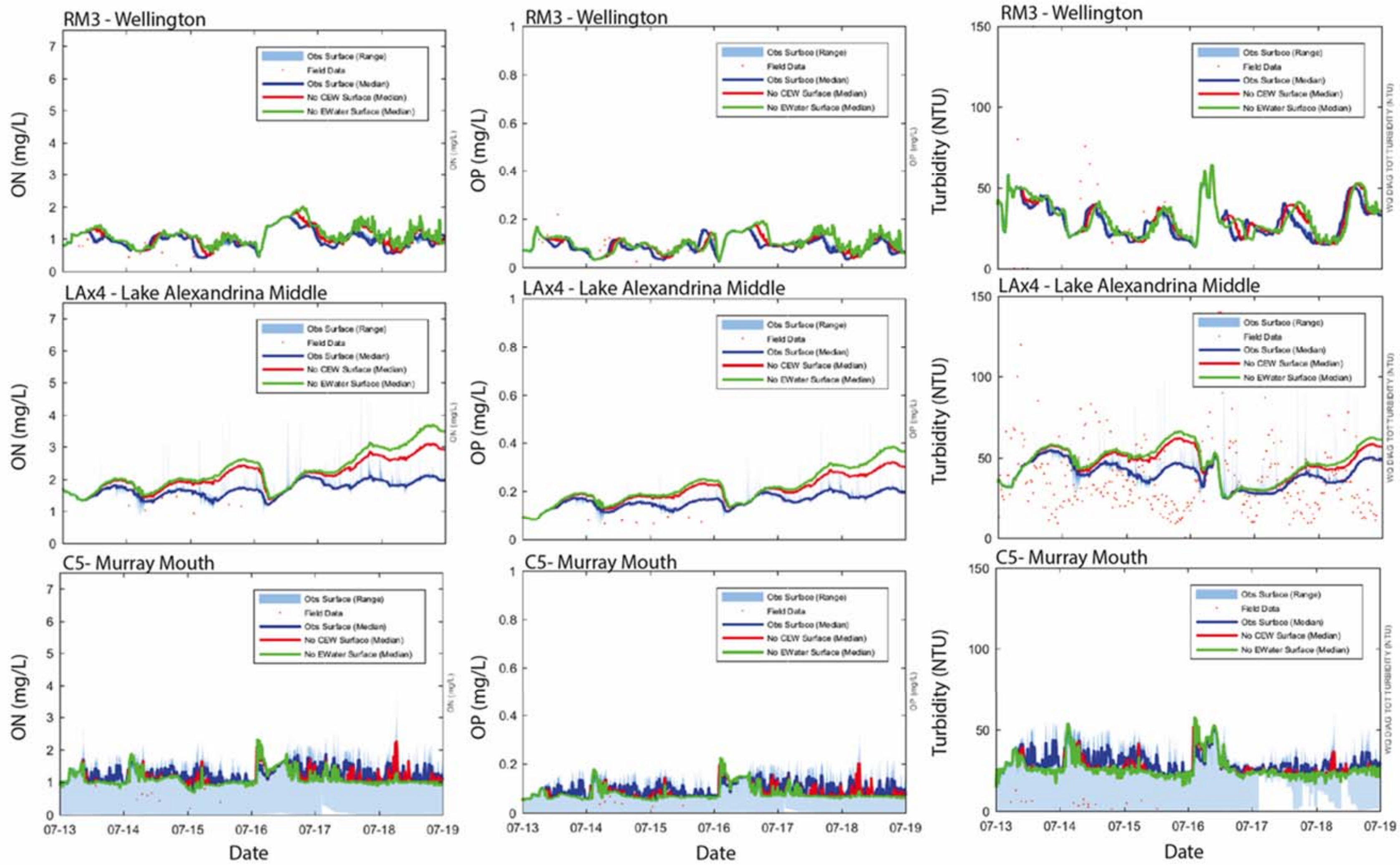

Figure D2. Daily concentrations of particulate organic nitrogen (left), particulate organic phosphorus (middle) and turbidity (right) at Wellington, Lake Alexandrina and the Murray Mouth under three different flow scenarios from 2013–2019.

Table D2. Microinvertebrate classification categories and their definitions for the purpose of this report.

Classification	Definition
Littoral	Organisms that prefer and are adapted to the region of the sublittoral zone up to the shore.
Littoral (facultatively pelagic)	Organisms that prefer and are adapted to the region of the sublittoral zone up to the shore however can also aptly survive in open water environments.
Pelagic	Organisms that prefer lentic open water environments.

Table D3. List of all of the rotifer, cladoceran and copepods species identified in 2014-15, 2015-16, 2016-17 and 2017-18 and their classification based on their preferred habitat. NR for Aust. = new record for Australia, NR for SA = new record for South Australia and n. sp. = new species.

Species identified	Group	Classification
Habrotrocha sp.	Rotifer	Littoral
Philodina alata **NR for Aust**	Rotifer	Littoral
Philodina sp.	Rotifer	Littoral
Rotaria neptunia	Rotifer	Littoral
Rotaria sp.	Rotifer	Littoral
indet. bdelloid [sm]	Rotifer	Littoral
indet. bdelloid [lg]	Rotifer	Littoral
Asplanchna cf. *brightwellii*	Rotifer	Pelagic
Asplanchna priodonta	Rotifer	Pelagic
Asplanchna sp.	Rotifer	Pelagic
Asplanchnopus sp.	Rotifer	Pelagic
Anuraeopsis coelata	Rotifer	Pelagic
Anuraeopsis fissa	Rotifer	Pelagic
Brachionus angularis	Rotifer	Pelagic
Brachionus bennini	Rotifer	Littoral (facultatively pelagic)
Brachionus bidens	Rotifer	Pelagic
Brachionus bidentatus	Rotifer	Littoral (facultatively pelagic)
Brachionus budapestinensis	Rotifer	Pelagic
Brachionus calyciflorus amphiceros	Rotifer	Pelagic
Brachionus calyciflorus s.l.	Rotifer	Pelagic
Brachionus caudatus personatus	Rotifer	Pelagic
Brachionus dichotomus reductus **NR for SA**	Rotifer	Pelagic
Brachionus diversicornis	Rotifer	Pelagic
Brachionus durgae **NR for Aust.**	Rotifer	Pelagic
Brachionus falcatus	Rotifer	Pelagic
Brachionus keikoa	Rotifer	Pelagic

Species identified	**Group**	**Classification**
Brachionus lyratus	Rotifer	Pelagic
Brachionus nilsoni	Rotifer	Pelagic
Brachionus novaezealandiae	Rotifer	Pelagic
Brachionus quadridentatus cluniorbicularis	Rotifer	Littoral (facultatively pelagic)
Brachionus quadridentatus s. str.	Rotifer	Pelagic
Brachionus rubens	Rotifer	Pelagic
Brachionus urceolaris	Rotifer	Pelagic
Brachionus **n. sp. [*angularis-lyratus* group]**	Rotifer	Pelagic
Brachionus sp.	Rotifer	Pelagic
Keratella americana **NR for Aust.**	Rotifer	Pelagic
Keratella australis	Rotifer	Pelagic
Keratella cochlearis	Rotifer	Pelagic
Keratella javana	Rotifer	Pelagic
Keratella lenzi **NR for SA**	Rotifer	Pelagic
Keratella procurva	Rotifer	Pelagic
Keratella quadrata	Rotifer	Pelagic
Keratella shieli	Rotifer	Pelagic
Keratella slacki	Rotifer	Pelagic
Keratella tecta **NR for SA**	Rotifer	Pelagic
Keratella tropica	Rotifer	Pelagic
Plationus patulus	Rotifer	Pelagic
Platyias quadricornis **NR for SA**	Rotifer	Pelagic
Collotheca pelagica **NR for SA**	Rotifer	Pelagic
Collotheca cf. *tenuilobata*	Rotifer	Pelagic
Collotheca sp.	Rotifer	Littoral
Conochilus dossuarius	Rotifer	Pelagic
Conochilus natans	Rotifer	Pelagic
cf. *Dicranophoroides* sp.	Rotifer	Littoral
cf. *Dicranophorus* sp.	Rotifer	Littoral
cf. *Encentrum* spp.	Rotifer	Littoral
Kostea wockei **NR for SA**	Rotifer	Littoral
Cyrtonia tuba **NR for SA**	Rotifer	Littoral
cf. *Epiphanes* sp.	Rotifer	Pelagic
cf. *Microcodides* sp.	Rotifer	Pelagic
Proalides tentaculatus	Rotifer	Pelagic
Proalides sp.	Rotifer	Pelagic
Beauchampiella eudactylota **NR for SA**	Rotifer	Littoral
Euchlanis sp.	Rotifer	Littoral
Ptygura sp.	Rotifer	Pelagic
flosculariid sp. [cf. *Sinanatherina*]	Rotifer	Littoral
Ascomorpha cf. *ovalis*	Rotifer	Pelagic
Ascomorpha saltans	Rotifer	Pelagic
Gastropus minor **NR for SA**	Rotifer	Pelagic
Hexarthra braziliensis **NR for Aust.**	Rotifer	Pelagic
Hexarthra intermedia	Rotifer	Pelagic

Species identified	**Group**	**Classification**
Hexarthra sp.	Rotifer	Pelagic
Lecane bulla	Rotifer	Littoral
Lecane 'bulloid' **n. sp.**	Rotifer	Littoral
Lecane closterocerca	Rotifer	Littoral
Lecane crepida	Rotifer	Littoral
Lecane curvicornis	Rotifer	Littoral
Lecane flexilis	Rotifer	Littoral
Lecane halsei **NR for SA**	Rotifer	Littoral
Lecane hamata	Rotifer	Littoral
Lecane nr *hamata* **?n. sp.**	Rotifer	Littoral
Lecane ludwigii	Rotifer	Littoral
Lecane luna	Rotifer	Littoral
Lecane lunaris	Rotifer	Littoral
Lecane obtusa	Rotifer	Littoral
Lecane signifera	Rotifer	Littoral
Lecane stenroosi	Rotifer	Littoral
Lecane ungulata	Rotifer	Littoral
Lecane (*s. str.*) sp.	Rotifer	Littoral
Lecane (*M.*) sp. **a**	Rotifer	Littoral
Lecane (*M.*) sp. **b**	Rotifer	Littoral
Colurella obtusa	Rotifer	Littoral
Colurella uncinata bicuspidata	Rotifer	Littoral
Colurella sp.	Rotifer	Littoral
Lepadella acuminata	Rotifer	Littoral
Lepadella patella	Rotifer	Littoral
Lepadella rhomboides	Rotifer	Littoral
Lepadella sp.	Rotifer	Littoral
Squatinella sp.	Rotifer	Littoral
Lindia sp.	Rotifer	Littoral
Lophocharis salpina	Rotifer	Littoral
cf. *Proales* sp.	Rotifer	Littoral
Cephalodella catellina	Rotifer	Littoral
Cephalodella forficula **NR for SA**	Rotifer	Littoral
Cephalodella gibba	Rotifer	Littoral
Cephalodella sp. a [v. sm]	Rotifer	Littoral
Cephalodella sp. b [med]	Rotifer	Littoral
Cephalodella sp. c [lg, elongate toes]	Rotifer	Littoral
Eosphora anthadis **NR for SA**	Rotifer	Littoral
Eosphora sp.	Rotifer	Littoral
Monommata sp.	Rotifer	Littoral
Notommata cf. *prodota* **NR for Aust.**	Rotifer	Littoral
Notommata spp.	Rotifer	Littoral
cf. *Resticula* sp. **[?n. sp.]**	Rotifer	Littoral
cf. *Taphrocampa* sp.	Rotifer	Littoral
indet. elong. notommatid	Rotifer	Littoral

Species identified	**Group**	**Classification**
Scaridium cf. *longicaudum*	Rotifer	Littoral
Polyarthra dolichoptera	Rotifer	Pelagic
Polyarthra vulgaris	Rotifer	Pelagic
Synchaeta oblonga	Rotifer	Pelagic
Synchaeta pectinata [med-lg, >100 µm]	Rotifer	Pelagic
Synchaeta **n. sp.**[tiny]	Rotifer	Pelagic
Pompholyx complanata	Rotifer	Pelagic
Testudinella patina	Rotifer	Pelagic
Trichocerca agnatha **NR for SA**	Rotifer	Littoral
Trichocerca bicristata	Rotifer	Littoral
Trichocerca bidens	Rotifer	Littoral
Trichocerca cf. *insignis* **NR for SA**	Rotifer	Littoral
Trichocerca pusilla	Rotifer	Littoral (facultatively pelagic)
Trichocerca rattus carinata **NR for SA** [was sp.a]	Rotifer	Littoral
Trichocerca similis	Rotifer	Littoral (facultatively pelagic)
Trichocerca similis grandis	Rotifer	Littoral (facultatively pelagic)
Trichocerca cf. *tigris*	Rotifer	Littoral
Trichocerca cf. *weberi*	Rotifer	Littoral
Trichocerca sp. **b** [tiny]	Rotifer	Littoral
Trichocerca sp. **c** [long toe, med]	Rotifer	Littoral
Trichocerca sp. **d** [gracile, med toe(s)]	Rotifer	Littoral
Trichocerca sp. **e** [sm bulb body, long toe]	Rotifer	Littoral
Trichocerca sp. **f** [oblate body, short toe]	Rotifer	Littoral
Trichocerca sp. **g** [small curved gracile, short toe]	Rotifer	Littoral
Trichocerca sp. **h** [robust, long toe]	Rotifer	Littoral
Macrochaetus sp. **NR for SA**	Rotifer	Littoral
Trichotria tetractis similis	Rotifer	Littoral
Filinia australiensis	Rotifer	Pelagic
Filinia brachiata	Rotifer	Pelagic
Filinia grandis	Rotifer	Pelagic
Filinia longiseta	Rotifer	Pelagic
Filinia opoliensis	Rotifer	Pelagic
Filinia passa	Rotifer	Pelagic
Filinia pejleri	Rotifer	Pelagic
Filinia terminalis	Rotifer	Pelagic
indet. 2-toed rotifer [sm]	Rotifer	Littoral
indet. glob. rotifer	Rotifer	Littoral
indet. plicate rotifer	Rotifer	Littoral
Bosmina meridionalis	Cladoceran	Pelagic
Armatalona macrocopa	Cladoceran	Littoral
Chydorus cf. *eurynotus*	Cladoceran	Littoral
Leberis diaphanus	Cladoceran	Littoral
Picripleuroxus quasidenticulatus	Cladoceran	Littoral
Pseudochydorus globosus	Cladoceran	Littoral

Species identified	**Group**	**Classification**
Pseudomonospilus diporus	Cladoceran	Littoral
indet. chydorid	Cladoceran	Littoral
Ceriodaphnia cornuta	Cladoceran	Pelagic
Ceriodaphnia sp. [non-*cornuta*]	Cladoceran	Pelagic
Daphnia carinata s.l.	Cladoceran	Pelagic
Daphnia galeata **NR for Aust.**	Cladoceran	Pelagic
Daphnia lumholtzi	Cladoceran	Pelagic
Daphnia sp. [non-lumh. late embryos]	Cladoceran	Pelagic
Simocephalus sp.	Cladoceran	Littoral
Ilyocryptus sp. [juv]	Cladoceran	Littoral
Macrothrix sp.	Cladoceran	Littoral
Moina cf. *australiensis*	Cladoceran	Pelagic
Moina micrura	Cladoceran	Pelagic
Moina cf. tenuicornis	Cladoceran	Pelagic
Neothrix sp.	Cladoceran	Littoral
Diaphanosoma excisum	Cladoceran	Pelagic
Boeckella triarticulata	Copepod	Pelagic
Calamoecia ampulla	Copepod	Pelagic
Calamoecia sp.	Copepod	Pelagic
Gladioferens sp. [female]	Copepod	Pelagic
calanoid copepodite	Copepod	Pelagic
calanoid nauplii	Copepod	Pelagic
Acanthocyclops cf. *vernalis* **NR for SA**	Copepod	Littoral
Australocyclops australis	Copepod	Littoral
Mesocyclops notius **NR for SA**	Copepod	Littoral
Microcyclops varicans	Copepod	Littoral
Thermocyclops sp.	Copepod	Littoral
indet subadult cyclopoid	Copepod	Littoral
cyclopoid copepodite	Copepod	Littoral
cyclopoid nauplii	Copepod	Littoral
indet. cyclopoid nauplius	Copepod	Littoral
indet. harpac.	Copepod	Littoral
harpac. copepodite	Copepod	Littoral
indet. copepod nauplius	Copepod	Littoral
Limnocythere sp.	Ostracod	Littoral
indet. ostracod [juv.]	Ostracod	Littoral

ACRONYMS

AHD	Australian Height Datum
CEW	Commonwealth environmental water
CEWO	Commonwealth Environmental Water Office
DEW	Department for Environment and Water
DOC	Dissolved organic carbon
ENP	Ecosystem net production
ER	Ecosystem respiration
GPP	Gross primary production
LMR	Lower Murray River (South Australian section of the Murray River).
LTIM	Long-Term Intervention Monitoring
M&E	Monitoring and Evaluation
MDB	Murray–Darling Basin
MDBA	Murray–Darling Basin Authority
NPL	Normal pool level
NSW DPIE	New South Wales Department of Planning, Industry and Environment
PSU	Practical salinity units
RMIF	River Murray Increased Flows
TL	Total length
TLM	The Living Murray
VEWH	Victorian Environmental Water Holder
YOY	Young-of-year

GLOSSARY

Allochthonous	Refers to foreign or outside sources. For example, organic matter of an allochthonous source is that which has been produced outside of the river channel, e.g. terrestrial or floodplain material.
Autochthonous	Refers to local sources. For example, organic matter of an autochthonous source is that which has been produced within the river channel.
Base flow	Flows that are confined to the low flow part of the river channel.
Biofilm	A collection of microorganisms (e.g. bacteria) attached as a 'film' on living (e.g. tree root) and non-living (e.g. wooden pylon) surfaces.
Flood or flooding	Refers to flows that are overbank. In South Australia, this is deemed to be above bankfull flow (45,000 ML/d).
Freshes (flow)	Flows greater than base flow but below bank level.
Epibenthic	Organisms living on the surface of sediment.
Epiphytic	Organisms that are attached to plants.
Heleoplankton	Plankton derived from billabongs and other floodplain still, generally-vegetated, waters.
In situ	Used to describe monitoring *in* the field.
Lentic	Refers to slower water velocities associated with 'pool water' habitat in highly regulated systems, typically median velocities of approximately ≤0.3 m/s.
Littoral	The margin along the bank of the river.
Lotic	Refers to flowing water, typically with median velocities of approximately >0.3 m/s.
Pulse (flow)	A description given to the shape of a hydrograph that is characterised by an increase in discharge, followed by a decrease in discharge, often of similar slope.
Recruitment (reproduction)	Refers to individuals passing the critical stages of early life (e.g. larval) and becoming juveniles in a population, described here as age 0+ years.
Respiration (ecosystem)	Ecosystem respiration is the measure of oxygen depletion in water by respiring animals.
RMIF	River Murray Increased Flows: a type of environmental water. Water entitlements recovered under the Snowy Water Initiative (established in 2002) via infrastructure upgrades and water purchase, which receive annual allocations and are used to supply environmental water to the Snowy River (Snowy River Increased Flows, SRIF) and River Murray (RMIF).
Primary productivity	The rate at which energy is converted to organic substances by autotrophs (e.g. algae and plants) during photosynthesis.
Southern connected Basin	The southern connected Basin is a network of the Murray River and all tributaries that flow into it between the Hume Dam and the sea. The Lower Darling (below Menindee Lakes) is considered part of the Southern Connected Basin, whilst all rivers upstream of Menindee Lakes are considered as the Northern Basin.
Unregulated flows	Unregulated flows occur when water in the system exceeds demands and are declared to be unregulated by the appropriate authority (source: http://www.bom.gov.au/water/awid/id-1026.shtml). They can be driven by substantial rainfall from upper tributaries, spills from headwork storages and rainfall rejection events.
Weir pool	Stretch of river between two weirs.