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Commonwealth Environmental Water Office
Long Term Intervention Monitoring Project
Goulburn River Selected Area
Scientific Report 2017–18

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1. Preamble

This *Scientific Report* is a companion volume to the *Summary Report* for the Goulburn River Long Term Intervention Monitoring (LTIM) Project (Webb et al. in review). The two documents complement each other, and repeat very little among them.

The Summary Report:

- Introduces the lower Goulburn River selected area and describes how it is treated for monitoring purposes
- Describes the Commonwealth environmental watering actions that occurred in the lower Goulburn River during 2017–18
- Provides the key outcomes for the five different monitoring disciplines undertaken: Hydraulic and Physical Habitat, Stream Metabolism, Macroinvertebrates, Vegetation, and Fish.
- Integrates these findings to update the conceptual model originally presented in the Monitoring and Evaluation Plan (Webb et al. 2018) that describes links among the different monitoring disciplines and the effects of flow upon them
- Considers the implications of the monitoring results for future management of Commonwealth Environmental Water

The Summary Report stands alone, in that it provides sufficient detail on the background and detail of the Goulburn River LTIM Project to be understood without reference to other documents.

This Scientific Report, in contrast, is not a stand-alone document, but is intended to be read alongside the Summary Report for those readers seeking more detail on different aspects of the Goulburn River LTIM Project than is possible within the space constraints for the Summary Report. In the sections below, the Scientific Report:

- Lists the specific monitoring sites in the Goulburn River LTIM Project and what monitoring activities are undertaken there
- Provides temporal summary of monitoring for 2017–18 versus what was planned
- Includes a detailed chapter on each of Physical Habitat, Stream Metabolism, Macroinvertebrates, Vegetation, and Fish. The chapters include:
    - Introduction, methods, results and discussion in the format of a standard report/paper
    - Evaluations of the area-specific monitoring questions being asked
    - Main findings from each of the monitoring disciplines for 2017–18 and how these build upon understanding developed in the first 3 years of the LTIM Project
- A report on our stakeholder communication activities for 2017–18

In this sense, the Scientific Report can be considered as a major appendix to the Summary Report.

2. Monitoring sites and 2017–18 monitoring

2.1 Sites

As described in the Summary Report (Webb et al. in review), the lower Goulburn River below Goulburn Weir is divided into two Zones for monitoring, with Zone 1 extending from Goulburn Weir to the confluence of the Broken River, and Zone 2 extending from there to the confluence with the Murray River.

Monitoring efforts are focused on Zone 2 to provide deeper understanding across a range of monitoring matters than would be possible if the program were spread evenly over the two zones. There are also several sites outside of the zones that provide important comparisons with results from within the Goulburn River. All monitoring sites are marked on Figure 1 of the Summary Report. Sites, apart from those where only hydrological data are collected, are detailed below (Table 2-1).

Table 2-1. LTIM monitoring sites in each zone and the monitoring activities undertaken at each site.

Site No.	Site Name	Adult Fish	Larval fish	Fish movement	2D Model	Bank Condition	Vegetation diversity	Stream metabolism	Macroinvertebrates
Zone 1 – Goulburn Weir to Broken River									
1	Moss Road				■				
2	Day Road							■	
3	Cable Hole			■					
4	Toolamba/Cemetery Bend		■	■					
5	Darcy's Track			■	■	■		■	
Zone 2 – Broken River to Murray River									
1	Shepparton Causeway	■							
2	Shepparton Weir			■					
3	Shepparton	■							
4	Zeerust	■							
5	Loch Garry Gauge	■	■	■	■	■	■	■	
6	Pogue Road	■							
7	Kotpuna	■		■					
8	McCoy's Bridge	■	■	■	■	■	■	■	■
9	Murrumbidgee Road	■							
10	Yambuna	■	■	■		■			
11	Sun Valley Road	■		■					
12	Stewarts Bridge	■							
13	Goulburn 0.3			■					
14	Murray Junction			■					
Outside of zones 1 & 2									
1	Central Avenue, Broken River								■
2	Murray 2			■					
3	Murray 1			■					
4	Murray -1			■					
5	Murray -3			■					

2.2 Monitoring in 2017–18

Following the disrupted monitoring schedule in 2016–17, caused by natural very high flows in spring including minor flooding on the lower Goulburn River, monitoring in 2017–18 proceeded much more according to plan (Table 2-2). All activities were implemented, and mostly in accordance with the original schedule. In addition, additional funding from the CEWO was provided in Autumn 2018 to undertake winter-focused monitoring in the final year of the LTIM Project. This meant that several activities were extended through to June (and beyond) on the schedule, which was not originally planned (Webb et al. 2018). The other main departure from planned activities was difficulties for summer monitoring caused by historically high Inter-Valley transfers over the period Jan-May 2018 (See Figure 1 – Summary Report). This affected sampling for bank condition and stream metabolism. A naturally high flow event in early December 2017 delayed vegetation sampling slightly and also meant that some transects and positions on transects could not be safely sampled.

The periods of monitoring for each activity are based upon the expected responses to flow variation, optimised for budgetary and logistic considerations. These reasons are given more fully in the recently updated Monitoring and Evaluation Plan (Webb et al. 2018). Updated Standard Operating Procedure (SOP) appendices are also included in that document to describe the additional monitoring described above. More detailed discussions of monitoring activities, how they differed from planned activities, results and discussion, are presented separately for each discipline in the following chapters.

Table 2-2. Schedule of planned and actual monitoring activities by month for 2017–18. D indicates planned/actual timing for downloading data from fish movement loggers; I indicates planned/actual deployment of artificial substrates for macroinvertebrate sampling, O indicates planned/actual retrieval of artificial substrates traps for macroinvertebrate sampling. C indicates additional sampling for biomass of crustaceans for macroinvertebrate sampling, first implemented in 2016–17, but now being continued for the remainder of the LTIM Project.

Monitoring activity	No of sites per Zone		Planned / Actual	Schedule of planned and actual activities in 2016-17											
	Zone 1	Zone 2		J	A	S	O	N	D	J	F	M	A	M	J
Adult Fish		10	Planned												
			Actual												
Fish Larvae	1	3	Planned												
			Actual												
Fish Movement	3	8	Planned			D			D			D			D
	+ 4 stations in the Murray River		Actual					D		D			D		
Vegetation Diversity		2	Planned												
			Actual												
Macroinvertebrates		1	Planned				I	O		I	O				
	+ 1 control site in the Broken River		Actual				I	O	C	IC	OC	C			
Stream Metabolism	2	2	Planned												
			Actual												
Bank Condition	2	2	Planned												
			Actual												
2D Hydraulic Model	2	2	Planned												
			Actual												

3. Physical habitat

3.1 Introduction

Hydraulic conditions, the state of river banks and sediment dynamics, greatly influence fish, vegetation and macroinvertebrate population dynamics. However, the relationships between discharge and river bank condition – such as erosion and deposition - are not well known. The physical habitat monitoring program has two elements for assessing the effects of Commonwealth environmental water: (i) linking flows to hydraulic habitat conditions for flora and fauna, and (ii) quantifying the role of flow in modifying bank condition, including erosion and deposition.

Hydraulic conditions specifically refer to metrics such as velocity and depth, rather than flow volume. Whilst, river managers often use flow volume as the main metric of study, it is the hydraulic conditions that influences the biota. For example, slackwater habitats are important nursery areas for fish larvae and juvenile fish, and are also areas of high productivity for zooplankton and macroinvertebrates. As such, flows that maximise the quality and quantity of slackwater habitats at critical periods are most likely to trigger a significant ecological response. Measuring changes in the distribution and quality of hydraulic habitats under different flow conditions is therefore important for determining whether specific flow management actions are providing the conditions required for an intended ecological outcome. Such information will improve the interpretation of ecological monitoring results, specifically the attribution of good ecological outcomes to the delivery of Commonwealth environmental water.

Hydraulic models are being used to quantify the relationships between discharge and ecologically relevant hydraulic metrics, to better understand the physical habitats in the Goulburn River. Model results can be used to produce discharge-habitat curves that allow us to predict the quality, quantity and distribution of specific hydraulic habitats under a wide range of flow magnitudes.

River banks influence the velocity of flow, depth of water, and provide the sediment conditions for ecosystem services such as habitat niches for vegetation. For example, a small amount of erosion can help streamside and instream vegetation become established, but excessive erosion can lead to sediment smothering of bed habitats, and harm to organisms therein. Quantifying the relationship between Commonwealth environmental water and bank condition can assist with identifying critical flow ranges to support specific aquatic biota and ecological processes.

3.2 Area specific evaluation questions

To determine the contribution of Commonwealth environmental water in selected areas, and to improve understanding of the relationship between specific watering actions and ecological objectives for assets, the following questions are being addressed. This information also forms the basis of Basin-scale evaluation – where area-level results are scaled up to the Basin level.

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
What did CEW contribute to the provision of productive habitat (e.g. slackwaters) for the recruitment, growth and survival of larval and juvenile fish?	The provision of baseflows and freshes in the 2017–18 season contributed to variation in the type and distribution of hydraulic habitat known to be of value to fish.	Both baseflows and freshes increase wetted perimeter, pool area and mean depth. Slackwaters (slow and shallow habitats) are high for lower discharges as the bed is one large slackwater. Slackwaters are minimised at discharges of ~5,000 ML/day and as such are decreased by higher events for that period, but are also increased again once benches are inundated. High velocities are considered to be important triggers for fish recruitment and migration and while average velocities tend to	Habitat relationships developed from two-dimensional hydraulic habitat models for four sites. These relationships will continue to be used to link biotic response and vegetation to water management and environmental flows.

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
		increases with increasing discharge, the higher (99th percentile) velocities are highest at lower flows <2,000 ML/day. The variable flows provided in the 2017–18 season provided considerable habitat variability.	
What did CEW contribute to the provision of diverse and productive macroinvertebrate habitats?	Baseflows and freshes, such as provided in the 2017–18 season, are known to provide habitat for macroinvertebrates.	Baseflows increase the wetted area of the channel bed, and freshes increase wetting on higher, often more productive features such as bars and benches. Freshes greatly increase the turnover of bed sediments; the area of sandy bed sediments mobilised is tripled when a fresh of 5,000 ML/day is provided, compared to a baseflow of 1,000 ML/day, and is important for flushing and renewal of bed sediments and habitats for macroinvertebrates.	Based on two-dimensional hydraulic modelling.
What did CEW contribute to inundating specific riparian vegetation zones and creating hydraulic habitats that favour the dispersal and deposition of plant seeds and propagules?	Freshes and variable flow levels, such as those achieved through flow management during the 2017–18 season, are known to increase opportunities for the dispersal and deposition of plant seeds and propagules.	Variable discharges and flow levels provide greater opportunities for the recruitment, transport and dispersal of seeds and propagules. High flow freshes, in particular, may transport the seeds and provide favourable conditions (wetting, low velocity) to encourage vegetation germination and growth on benches and banks. High velocities may also be an important factor in the creation of niches for seed deposition. Outcomes require confirmation by coordinating hydraulic results with vegetation analyses and the coordinated collection and analysis of sediment and seeds as is currently underway in the Goulburn River.	Hydraulic models have demonstrated changes in velocities at banks where vegetation is sampled. Mud drapes observed (and measured) following higher discharges have been observed to be zones of vegetation growth. Further coordination of hydraulic results and vegetation, and results from the use of turf mats for the collection and analysis of seeds and sediments, will confirm relationships.
How does CEW affect bank erosion and deposition?	Magnitude, frequency and duration of flows were all appropriate to prevent excessive rates of riverbank erosion and to also allow for deposition.	Erosion and deposition are most closely related to the duration of flows, and environmental flows influence this activity, but the effect is not significant. Levels of erosion are slightly higher than the levels of aggradation/deposition, but are also related to the program targeting sensitive banks to ensure relationships can be measured. Some minor mass failure (bank slumping) was observed following notching related to IVT flows. Episodic changes observed are not expected to be outside natural levels of variation, and where erosion does occur this was observed to provide niches for organic matter and vegetation establishment.	Bank condition is based on quantitative measurements of bank erosion using erosion pins. At each site, erosion pins located at varying levels and locations, are re-measured pre/post events to assess bank change (levels of erosion or deposition). Statistical models compared predicted erosion/deposition under actual flow regime and one from which environmental flows had been removed.

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
How does the amount of river bank erosion affect vegetation responses to environmental water delivery?	Inundation frequency was appropriate to encourage lower bank vegetation, velocities at banks were not excessive, and mud drapes resulted following drawdown.	Whilst vegetation response has not been formally incorporated into the bank condition assessment at this stage, the flows delivered maintained appropriate rates of erosion and deposition and were found, in some cases, to encourage vegetation establishment. Low rates of recession commonly left ‘mud drapes’, particularly on the lower banks, providing suitable substrate for the germination of a range of plants.	Assessment of hydrologic conditions, qualitative assessments of erosion mechanisms, and observations (including repeat photographs) have enabled an assessment of bank condition and the potential for vegetation establishment and this will be quantified by coordinating the bank monitoring and vegetation results.

3.3 Main findings from the physical habitat monitoring program

3.3.1 Hydraulic habitat findings from 2017–18 reinforce previous findings

- Hydraulic habitat relationships can provide specific flow targets and ranges to refine flow planning and adjust flow conditions to suit targeted outcomes and minimise potential risks, which allows further targeted research that informs the process.

- Bed mobilisation demonstrates that increases in discharge significantly increase the potential for bed substrate turnover. This ‘disturbance’ is important for refreshing sediment, promoting the processing of organic material and nutrients and providing a mosaic of benthic habitats for a range of biota, including macroinvertebrates, algae and macrophytes.

- Slow and shallow ‘slackwater’ area (where depth is less than 0.5 m and velocity is less than 0.05 m/s) is increased from zero flow as the bed is inundated. The relationship between discharge and slackwater area and distribution varies across sites, but area tends to be maximised at flows in the range of 1,000-5,000 ML/day (depending on individual sites). As flows increase further the total area of slackwater decreases and mean patch size decreases, however, the number of individual patches increases (i.e. higher discharges result in more but smaller slackwater patches). The optimal slackwater patch size is not known and could be investigated further.

- The relationship between velocity and flow rate depends greatly on the metric selected, thus the metrics must be specifically defined relative to the hydraulic habitat of interest. For example, mean velocity increases with flow rate (for all sites). Maximum velocity, however, decreases for increasing flow rate until approximately 2,000 ML/day, then gradually increases for increasing flows beyond this. The distribution of velocities across the channel also varies with discharge. For example, velocity on the banks tends to be lower than in the channel. Velocities greater than 0.3 m/s may have the capability to influence vegetation and may assist with explaining changes to bank vegetation. This also appears to be an important velocity threshold for golden perch spawning. The modelling suggests that rates of change in velocity are greatest for lower flows, less than ~2,000 ML/day.

- Bench inundation generally increases to a maximum between 1,000-5,000 ML/day and as such the vegetation and sediment deposition on benches is dependent on freshes.

Hydraulic conditions (such as velocity, depths and bed substrate turnover) for specific biota can be manipulated through flow management. For example, adding a fresh of 5,000 ML/day to baseflow can triple substrate turnover, reducing sediment smothering and increasing bed sediment diversity. The key element to the strategic use of hydraulic conditions as a tool for flow management is in understanding the preferred conditions for biota and the timing of these requirements. The quantification of habitat relative to discharge is providing opportunities for the water managers to understand the potential implications of particular discharge and tailor flow events accordingly. The mechanistic links between hydraulic habitat and biota will be further developed as the program proceeds.

3.3.2 Bank condition findings from 2017–18 reinforce previous findings

- The main finding reinforced by this year's program is that the hot season (summer/autumn) leads to a greater likelihood of erosion than the cold season (winter/spring). Observations suggest the sub-aerial drying (desiccation) that occurs in the warmer periods makes the clay-rich banks crack and leaves them prone to erosion once water levels rise. This highlights that preparation of banks may be more important than the flow event that is associated with erosion. It also leads to an important role of vegetation in shading banks from direct sun.

- Notching of riverbanks, whereby a visible line of erosion is associated with the flow level maintained in the previous event, was more evident following the 2018 IVT flow period, despite the variability in flow that was achieved.

- Bank erosion and deposition is highly variable with time, with a single point on the bank changing from erosion to deposition with subsequent flow events. Erosion also varies spatially, both along the riverbank and with elevation, often over small spatial scales of centimetres to metres.

- Monitoring shows there is marginally more erosion than deposition, but this may also be an artefact of targeting sensitive banks to better understand relationships between flow and bank condition. Significant erosion (>30 mm) is not common.

- The likelihood of erosion is most strongly linked to the duration of inundation. The longer the duration of bank inundation, the higher the likelihood of minor erosion (< 30 mm). High rates of drawdown and freshes/high flows following dry periods in summer (the hot season) also marginally increase the probability of minor erosion occurring, but the increases are not statistically significant.

- There appears to be no influence of peak discharge or flow volume on bank erosion.

- Freshes that inundate sediment after a dry period in summer were hypothesised to result in higher likelihood of erosion (compared to freshes in spring) but the results do not support this, suggesting that if anything winter/spring environmental flows have more influence on erosion.

- There is a slightly higher probability of minor bank erosion at lower bank elevations with increased inundation due to environmental flows (~10% increase). This is not surprising considering increased frequency of inundation of the lower banks and the relationship to inundation. The trend is less pronounced for significant erosion. Deposition is also increased due to inundation of the lower banks by environmental flows (i.e. erosion occurs during the rising flow and deposition occurs on the descending flow, so the net impact is small and variable).

3.3.3 The main findings from 2017–18 monitoring can be summarised as:

- Current flow management approaches in the Goulburn River are not leading to excessive riverbank erosion and current considerations for event management should be continued (these are explored in the following points):

    - Maintain variability in flows and water levels to maintain bank wetting at varying levels to avoid bank 'notching'. It was found that notching during the IVT flows led to some localised bank slumping on the recession of the following flows;

    - Maintain 'piggy backing' on tributary inflows to ensure sediment from tributaries is transported and deposited at higher levels in the channel (bars, benches, upper banks) during high flow freshes;

    - Manage maximum rates of flow recession within current levels to avoid bank surcharging and erosion, and allow mud drapes to develop. Mud drapes on banks have been associated with vegetation growth, and the potential for increasing mud drapes is currently being investigated as a project for the CEWO, and;

- Continue the modification of flow management as a collaborative effort between the researchers and water managers.

3.4 Physical Habitat Sites

Four sites are used for the hydraulic habitat and bank condition monitoring (Table 3-1). Moss Road is only used for hydraulic habitat monitoring, and Yambuna Bridge is only used for bank condition monitoring. This variation is to maximise the value of the specific questions being posed for each of these monitoring programs.

The methods for monitoring hydraulic habitat and bank condition are described in detail in the Standard Operating Procedures (SOPs) (Webb et al. 2018). Hydraulic data, model development and verification is described in detail in the *Commonwealth Environmental Water Office Long Term Intervention Monitoring Project Goulburn River Selected Area evaluation report 2014–15* (Webb et al. 2016). The hydraulic modelling methods are summarised here. Methods for bank condition monitoring were also described in detail in the 2014–15 report and are therefore also summarised here. Statistical analyses have been performed on data collected for the entire program and therefore results are for the four-year period. Observations from the 2017–18 period are described here to highlight changes in this water year.

3.5 Hydraulic habitat

3.5.1 Hydraulic habitat model development

Hydraulic habitat (i.e. velocity, depth etc.) is assessed using a hydraulic model that can be used to characterise hydraulic conditions for particular discharges. The model is two-dimensional such that velocity is resolved in both x and y directions, and not averaged across the channel (as with the one-dimensional models used for basic water level assessment). The difference in the model for the 2017–18 period is the extension of the flow-habitat curves to consider higher discharges for the Darcy's Track and McCoys Bridge sites (15,000, 18,000 and 20,000 ML/day). This was based on a request from the GBCMA due to interest in higher level discharges.

The model requires bed topography as an input, developed using two approaches for above and below water level. Surface topography was obtained from LiDAR (provided by the GBCMA). For the inundated sections bathymetry was captured by Austral Research using a remote-controlled Sonar boat (Z-Boat 1800, Figure 3-1, left). These data points are joined in GIS to produce a topographic surface (Figure 3-2).

For verification purposes field velocities were measured using an Acoustic Doppler Current Profiler (ADCP) at a range of discharges for model verification (Figure 3-1, right).

3.5.2 Elevation data verification

The same procedure for model development and verification is followed for each of the four sites. For brevity the descriptions here of development, verification and results are presented for one site, Moss Rd.

The bathymetry XYZ file was triangulated in ArcGIS and converted to a 1 m resolution grid. The bathymetry TIN (triangulated irregular network) was compared to the LiDAR grid in the areas where they overlapped. The area of overlap was based on visual assessment and clipping out of water surface from LiDAR.

The mean difference between the two datasets was 0.22 m (LiDAR higher than bathymetry) and the standard deviation of differences was 0.36 m, indicating noise in one or both datasets. The median difference was 0.17 m.

Table 3-1. Goulburn River LTIM physical habitat monitoring sites for physical habitat (hydraulic modelling) and bank condition.

	Site (Component)	Coordinates	Image
1	Moss Road (physical habitat)	E 337458.08 N 5936838.35	
2	Darcy's Track (physical habitat and bank condition)	E 351721.99, N 5966032.91	
3	Loch Garry (physical habitat and bank condition)	E 345932.83 N 5987637.56	
4	McCoy's Bridge (physical habitat and bank condition)	E 330801.78 N 5994732.86	
5	Yambuna Bridge (bank condition)	E 360741.50 N 1450010.78	

Figure 3-1. Instruments used to collect field data for development and verification of the hydraulic model: (left) Sonar bathymetric survey boat, (right) Acoustic Doppler Current Profiler (tethered to a rope to obtain velocities across fixed cross sections).

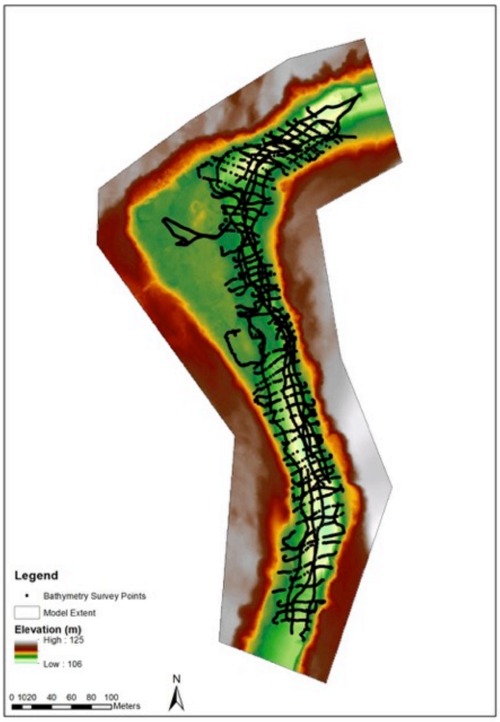


Figure 3-2. Topography used to develop the hydraulic model for Moss Rd based on LiDAR and bathymetric survey. The main channel (represented here in green) has the path of the bathymetric survey overlain in black to demonstrate coverage. This includes some verification runs of the boat into the backwater section (already covered by LiDAR).

3.5.3 Mesh Setup, Boundary Conditions and Roughness

The 1 m LiDAR/bathymetry grid was exported to text format for input to the River 2D program. The R2DMesh program (Figure 3-3) was used to create a triangular mesh of the following approximate resolution:

- In-channel (bank to bank): 2 m
- Floodplain: 8 m
- Transition: 4 m

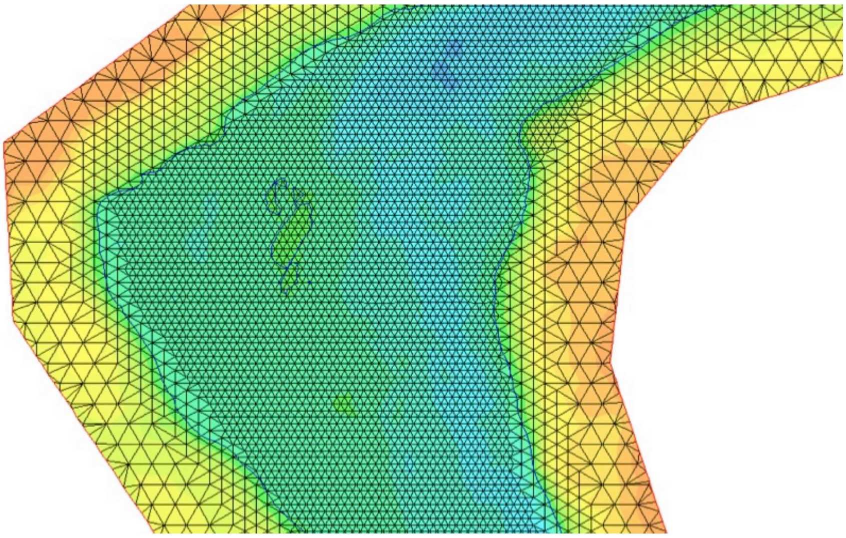


Figure 3-3. Example of computational mesh resolution and setup for Moss Road. Greater detail (higher resolution) is provided within the channel to capture small-scale hydraulic variation on the bed of the channel and for lower velocities.

The upstream boundary condition was set to a constant inflow. The downstream boundary condition was set to a constant water level boundary.

River2D requires the input of a roughness height in metres. A variable roughness height was used (Figure 3-4) for different bed cover types with the following values:

- Background: 0.2 m
- Rougher channel adjacent to large bar: 0.3 m
- Wood not in bathymetry: 1 m
- Sparse Riparian Vegetation: 0.5 m
- Moderate Riparian Vegetation: 0.8 m
- Dense Riparian Vegetation and Wood: 1.0 m

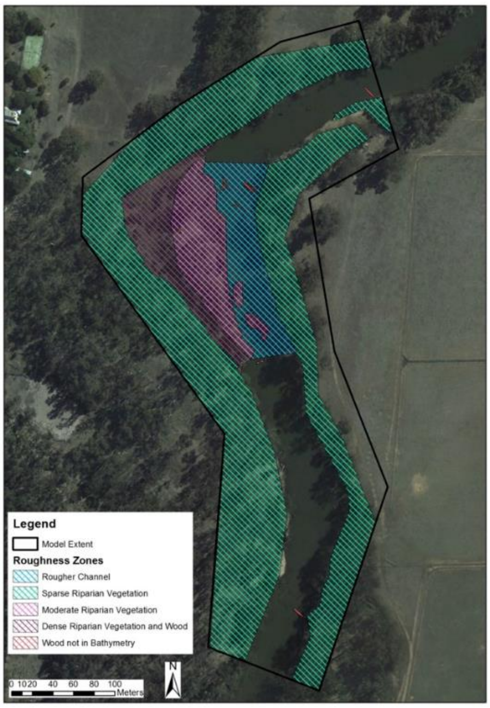


Figure 3-4. Roughness zones for Moss Road

3.5.4 Calibration

Two calibration events were available (Table 3-2). The events were run through the model using the average flow from the ADCP profiles, which were considered more representative at the site than the gauged data at Murchison. The ADCP flows were internally consistent (9–10 m^3/s for the low flow event and 33–40 m^3/s for the high flow event) and reasonably consistent with the gauged flow (0–10% lower for the low flow event and 13–28% lower for the high flow event). The tailwater was calculated from interpolation of the design tailwater levels.

Table 3-2. Moss Road calibration data.

Date	Average flow from ADCP data (m^3/s)	Gauged flow at Murchison (m^3/s)	Observed data	Adopted flow (m^3/s)	Adopted tailwater (m AHD)
12/6/2015	9.4	10.0	ADCP velocity (x, y, magnitude and direction) at 5 sections	9.4	112.8
25/6/2015	37	46	ADCP velocity (x, y, magnitude and direction) at 5 sections	37	111.6

Velocity magnitude results were extracted at each ADCP observation point for comparison. Average differences for each section, as well as standard deviations of the differences and maximum differences, are given in Table 3-3. Modelled velocities were generally within +/- 0.1 m with no apparent bias.

Table 3-3. Moss Road calibration results.

Date	Section	Average difference (modelled – measured) (m/s)	St. dev. of differences (m/s)	Max difference (m/s)
12/6/2015	4	-0.01	0.08	-0.17
	6	0.008	0.08	-0.16
	8	-0.04	0.14	-0.32
	9	-0.04	0.04	-0.15
	10	-0.02	0.02	-0.12
	Total	-0.02	0.08	-0.32
25/6/2015	4	0.03	0.06	0.18
	6	0.05	0.14	0.32
	8	-0.03	0.19	-0.95
	9	0.005	0.05	0.12
	10	-0.01	0.06	-0.15
	Total	0.01	0.12	-0.95

For the low flow event, a scatter plot showing observed and modelled velocity magnitude values for each section is given in Figure 3-5, and a plot showing the velocity differences spatially is shown in Figure 3-6. The same plots for the high flow events are given in Figure 3-7 and Figure 3-8. The observed velocity profile may have been produced by a local but temporary blockage. Localised obstructions (e.g. wood) may be the cause of this variability. Rather than make arbitrary changes to the topography, the calibration was accepted as is, noting that results at the channel margins at low flows may have higher uncertainty than elsewhere.

For the high flow event, Section 8 again had some significant discrepancies between observed and modelled velocities. At three points in particular observed velocities were underestimated by 0.6–0.95 m/s by the model. Given these observed velocities were outside the bounds of any other measured velocities in this event, and much higher than adjacent velocities on the same section, this was attributed to instrument or measurement error which is common in shallow environments.

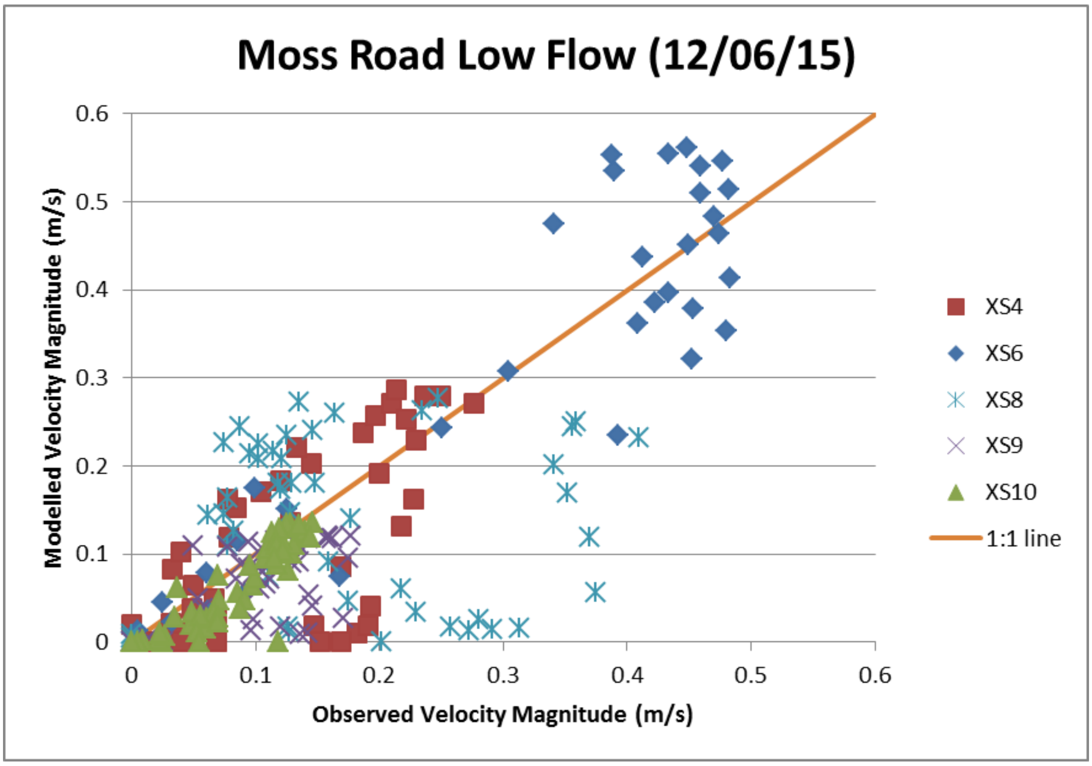


Figure 3-5. Calibration results (velocity comparison) for Moss Road low flow event (12/06/15)

Figure 3-6. Calibration results (velocity difference) for Moss Road low flow event (12/06/15)

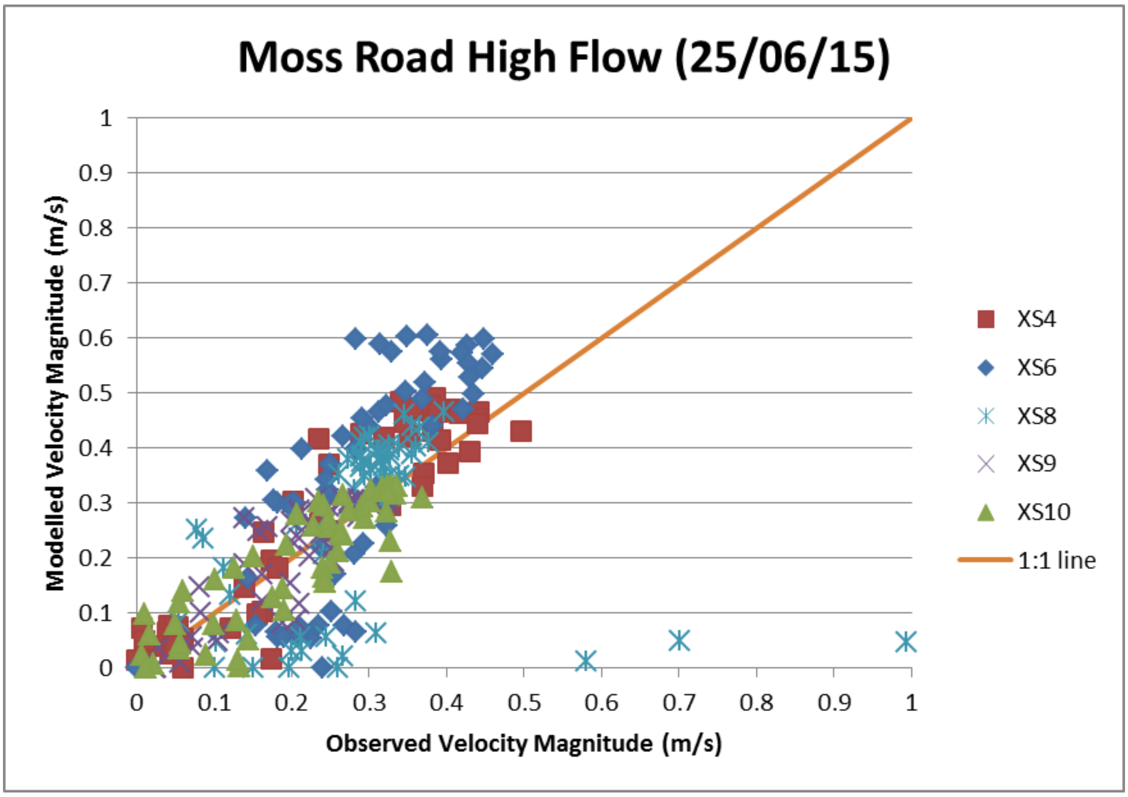


Figure 3-7. Calibration results (velocity comparison) for Moss Road high flow event (25/06/15)

Figure 3-8. Calibration results (velocity difference) for Moss Road high flow event (25/06/15)

3.5.5 Hydraulic model outputs

The following outputs were extracted from a 1m grid using depth, velocity (example Figure 3-9) and shear velocity (bed shear stress $\tau_b = \rho u_*^2 = 1000u_*^2$):

- Mean velocity
- 99th percentile velocity
- Wetted area
- Bench inundation - % of bench area inundated (bench area definition was undertaken manually in ArcGIS using the digital elevation model)
- Sediment mobilisation (proportion of bed with 1mm/2mm sediment mobilised, based on shear stress thresholds using a dimensionless Shields parameter of 0.06, specifically: 0.97 N/m^2 for 1 mm sediments and 1.94 N/m^2 for 2 mm sediments).

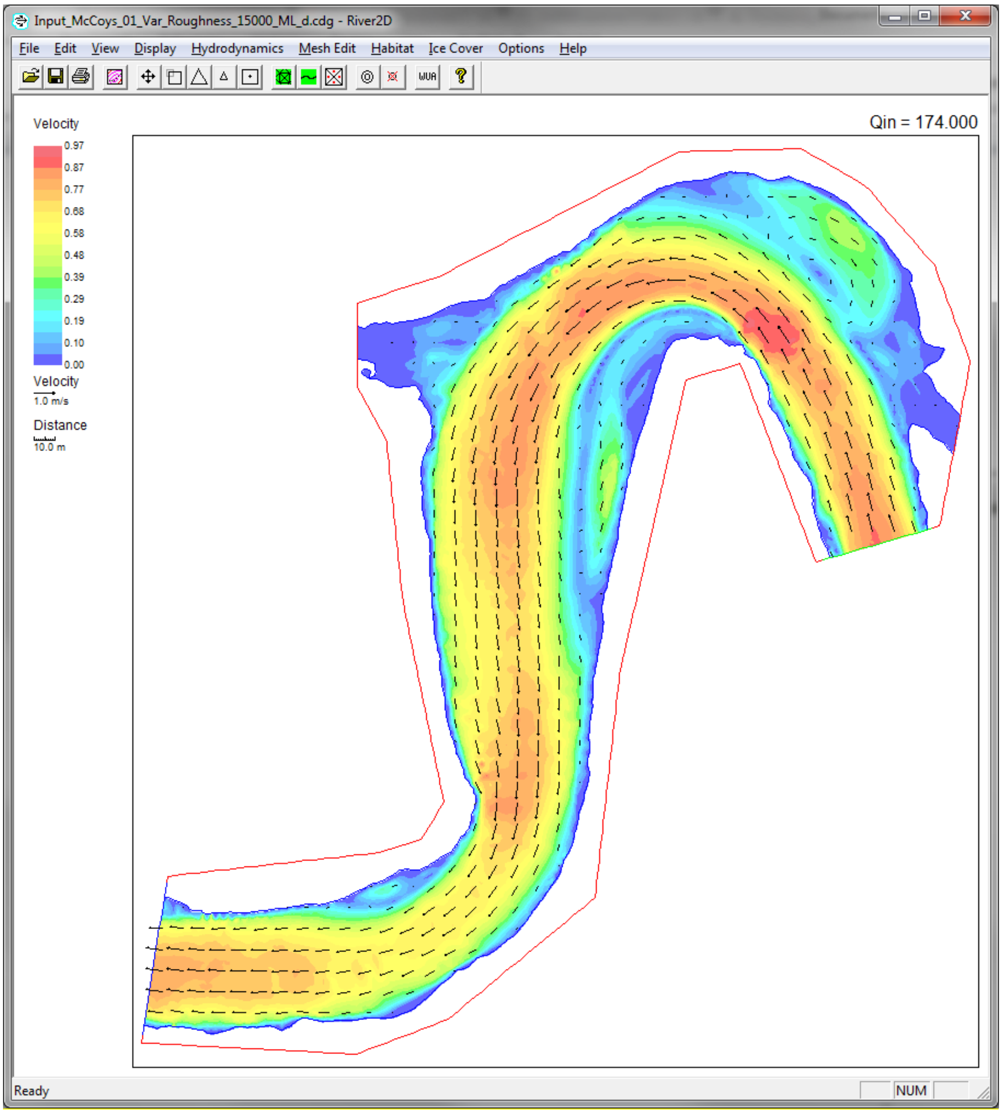


Figure 3-9. Velocity results for McCoys Bridge at a high flow of 15,000 ML/day.

3.5.6 Results Hydraulic Habitat

Results were extracted for a range of steady state simulations, from a low flow of 300 ML/day up to beyond bankfull flow with a flow of 20,000 ML/day (extended beyond 12,000 ML/day for Darcys Track and McCoys Bridge, with results for the latter shown in Table 3-4). Examples of relationships are shown for Moss Road and McCoys Bridge.

As discharge increases total wetted area and the area of pools (deeper than 1 m and 1.5 m) increases most dramatically for discharges between 4-5,000 ML/day, then to a lesser extent thereafter (Figure 3-10).

The area of slackwater habitat (Figure 3-11), where depth is less than 0.5 m and velocity is less than 0.05 m/s, generally increases to a maximum for very low discharges then decreases to a minimum at approximately 4-6,000 ML/day. Then the area of slackwater habitat varies with minor increases once higher-level benches are inundated. The number of slackwater patches decreases as discharge increases (Figure 3-12).

Average velocity generally increases with increasing discharge (Figure 3-13), but this does not necessarily represent the velocities of interest to biota. Relationships between fish spawning and velocities were developed by the team with input from Wayne Koster and analysis by Angus Webb. Consideration of fish triggers, and the relationship between fish movement and velocities, led to investigation of maximal velocities. Maximum velocities (averaged over the reach) and high velocities (the 99th percentile, to remove the potential for extreme values) are presented in (Figure 3-14). This demonstrates that maximum velocities can be very high at low average velocities (<1000 ML/day), where some very localised maximum velocities can occur. High average velocities tend to moderate this pattern. The rate of change in velocity is also considered as a potential trigger for fish movement (Figure 3-15). For example, at a flow rate of 6,500 ML/day, an increase of 100 ML/day would produce an increase in average velocity of 0.0024 m/s over the same time period. Rate of change in velocity is greater for lower discharges, i.e. there is a relatively larger shift in velocity for a change in low discharges.

For vegetation there was consideration of impacts by maximum velocities. Maximum velocity at vegetation transects was developed by extracting velocity at the specific locations where vegetation samples were taken (Figure 3-16).

Disturbance of substrates and the potential for particular discharges to mobilise bed sediments is based on shear stress. Figure 3-17 demonstrates that coarse-grained sediments such as gravels (>2mm) require discharges of more than 2,500 ML/day before significant bed movement occurs. Medium-grained sized sands (>1mm) are mobilised readily and significantly larger areas of the bed are mobilised as discharge increases up to 5,000 ML/day. The area of sandy bed sediments mobilised is tripled when a fresh of 5,000 ML/day is provided, compared to a baseflow of 1,000 ML/day.

Bench inundation dramatically increases beyond 1,000 ML/day and reaches a maximum near 4,000 ML/day for McCoys Bridge (Figure 3-18). Bench inundation is highly dependent on channel morphology and is often associated with one or two benches in each reach. However, among the reaches assessed the elevations of benches occur at surprisingly consistent levels.

Table 3-4. McCoys Bridge habitat area results

Flow (ML / day)	Flow (m^3/s)	Mean velocity (m/s)	Wetted area (m^2)	Area of pools > 1.0 m (m^2)	Area of pools > 1.5 m (m^2)	Area of slackwater habitat (D < 0.5 m, V < 0.05 m/s) (m^2)	No. patches slackwater habitat	Mean patch size of slackwater habitat (m2)	Area bed shear > 0.97 N/m2 (1 mm sediment mobilised)	Area bed shear > 1.94 N/m2 (2 mm sediment mobilised)	Bench area inundated (m2)	Change in velocity per ML/day change in flow (m/s/ML/day)	High Velocity (99%) (m/s)	Max Depth (m)
300	3	0.10	24,875	14,048	9,769	3,659	93	39	2,082	1,616	-	0.000320	0.68	5.76
500	6	0.13	26,741	15,179	10,804	3,456	101	34	3,104	2,193	-	0.000184	0.78	5.96
1,000	12	0.20	29,662	17,752	13,723	2,756	130	21	6,806	3,306	113	0.000127	0.60	6.34
2,000	23	0.24	33,829	24,461	17,993	3,066	183	17	10,422	2,609	1,701	0.000041	0.51	6.88
3,000	35	0.27	36,988	29,541	23,902	2,004	223	9	14,777	3,613	3,330	0.000037	0.56	7.35
4,000	46	0.31	38,408	32,475	28,490	1,450	249	6	18,910	4,541	3,692	0.000032	0.60	7.73
5,000	58	0.33	39,601	35,588	31,401	1,292	273	5	22,021	5,781	3,809	0.000028	0.63	8.10
6,000	69	0.36	40,662	37,290	34,215	1,267	290	4	23,936	8,149	3,846	0.000023	0.65	8.42
7,000	81	0.38	41,863	38,455	36,484	1,320	284	5	25,750	10,312	3,855	0.000020	0.66	8.75
8,000	93	0.39	43,058	39,495	37,875	1,390	310	4	26,972	13,100	3,855	0.000015	0.69	9.07
10,000	116	0.42	45,636	41,468	39,706	1,787	287	6	29,022	18,397	3,855	0.000014	0.73	9.64
12,000	139	0.44	48,098	43,420	41,536	1,794	270	7	30,056	20,958	3,855	0.000011	0.79	10.17
15,000	174	0.48	51,096	46,397	43,969	1,647			32,517	24,558	3,855	0.000012	0.87	10.80
18,000	208	0.52	53,226	48,757	46,412	1,531			34,492	27,864	3,855	0.000013	0.95	11.30
20,000	231	0.54	54,495	50,277	47,803	1,465			35,481	29,256	3,855	0.000011	0.99	11.60

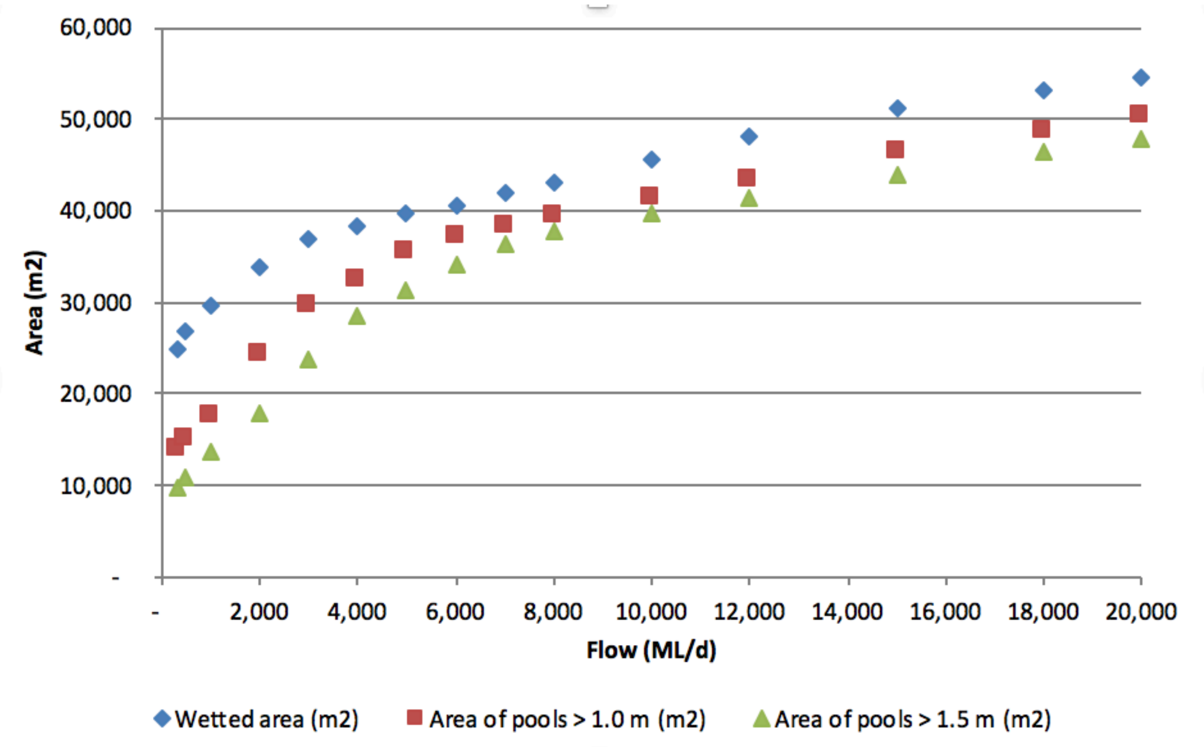


Figure 3-10. Results (wetted area and area of pools) for McCoys Bridge.

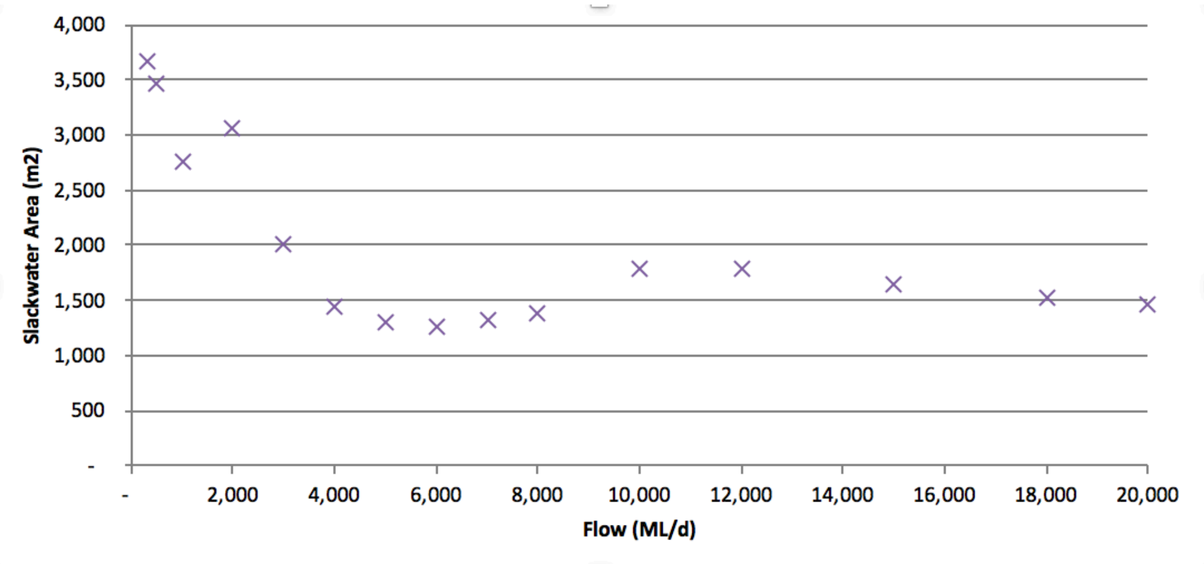


Figure 3-11. Results (area of slackwater habitat) for McCoys Bridge.

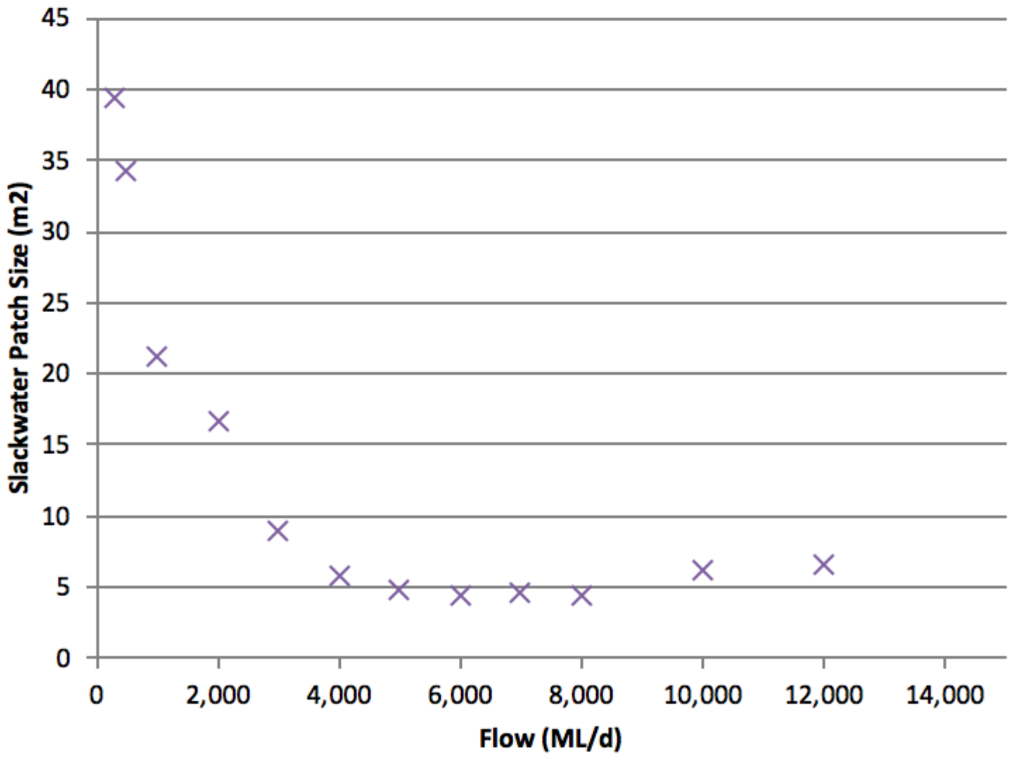


Figure 3-12. Results (mean patch size of slackwater habitat) for McCoys Bridge

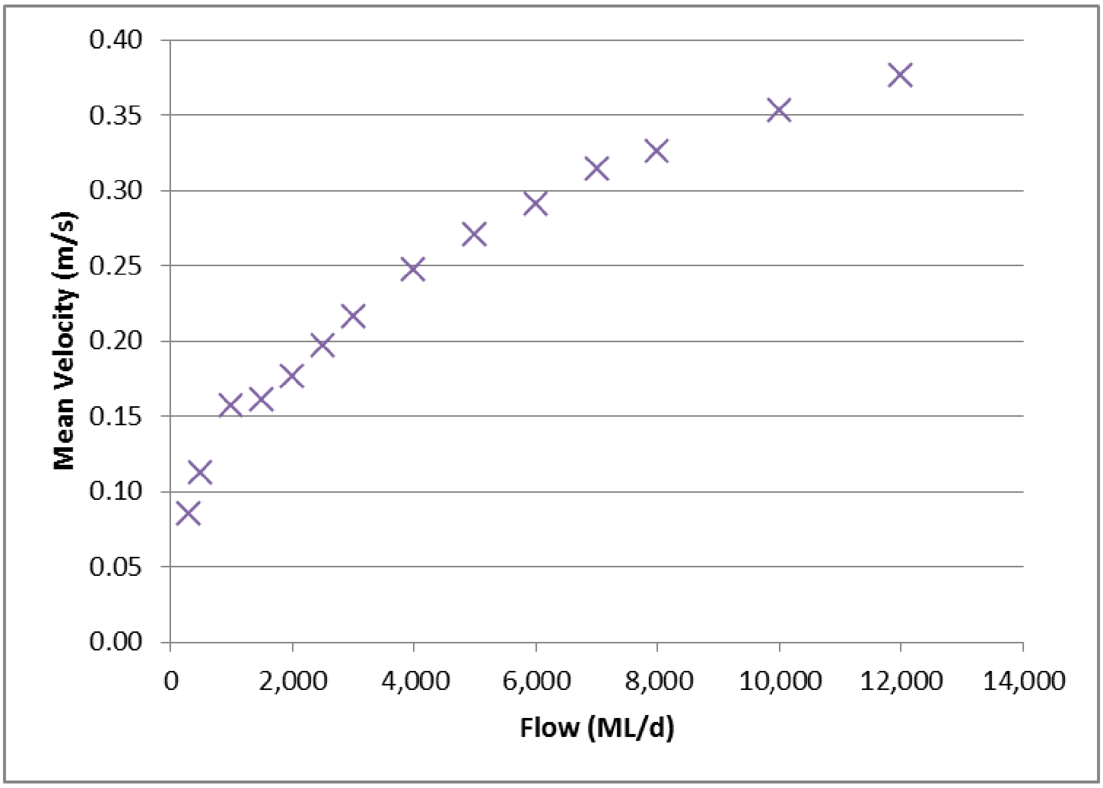


Figure 3-13. Results (mean velocity) for Moss Road

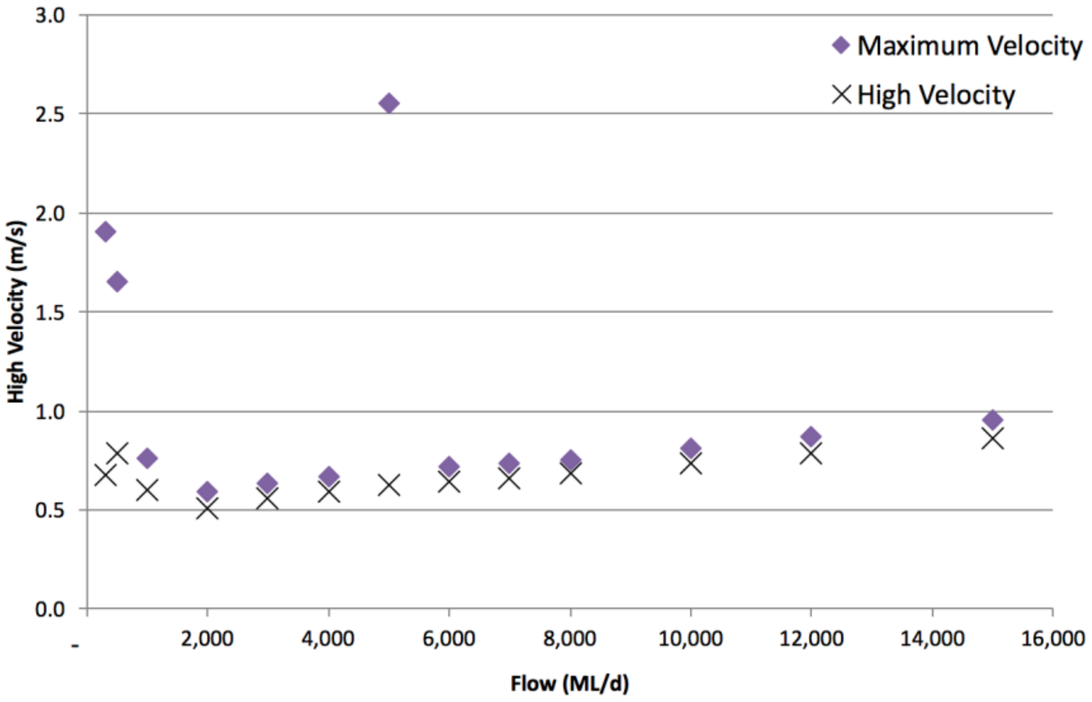


Figure 3-14. Results for maximum velocity (purple solid) and high velocity (crosses) relative to discharge for McCoys Bridge

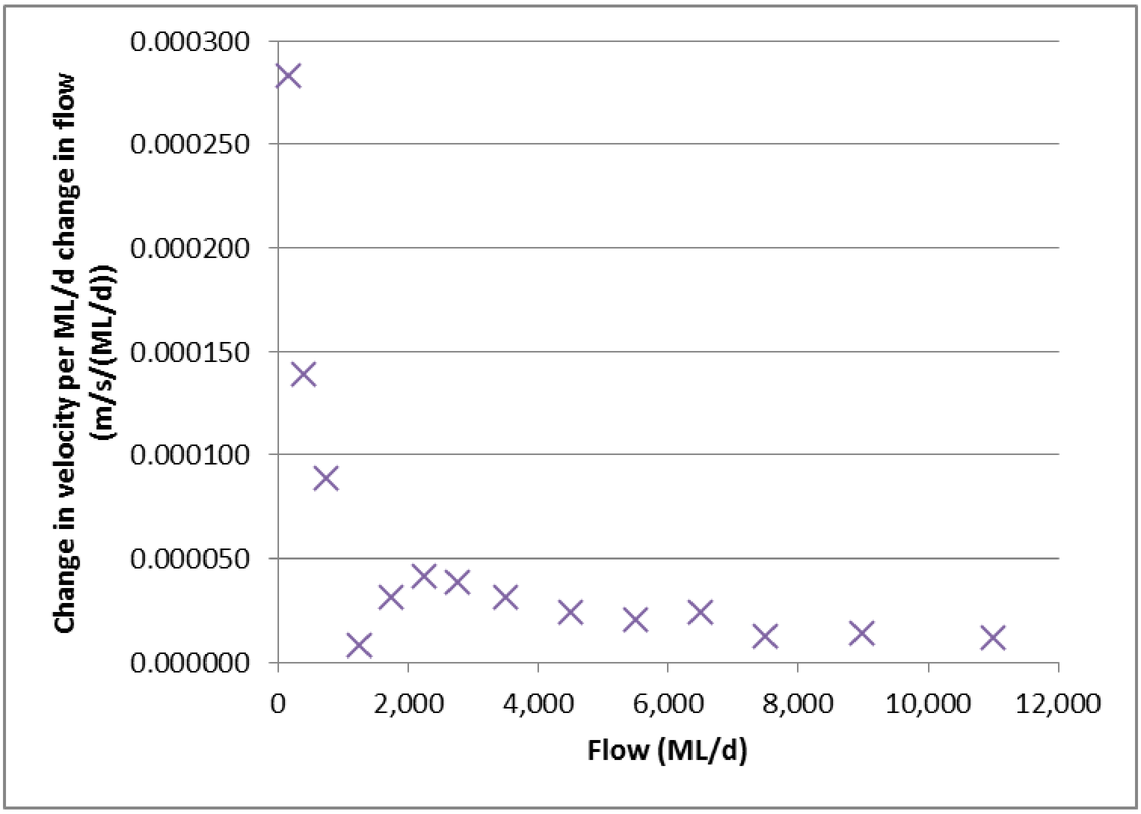


Figure 3-15. Results (velocity rate of change with flow) for Moss Road

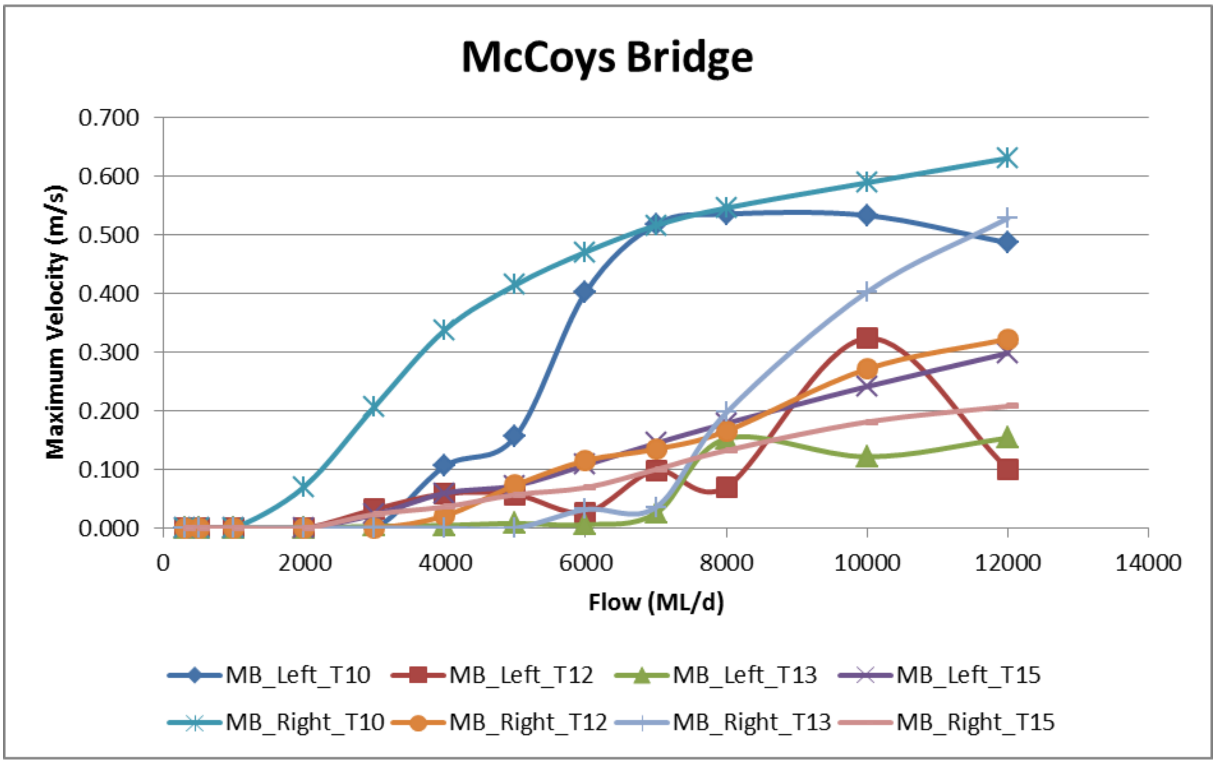


Figure 3-16. Maximum velocity at vegetation transects for McCoy's Bridge

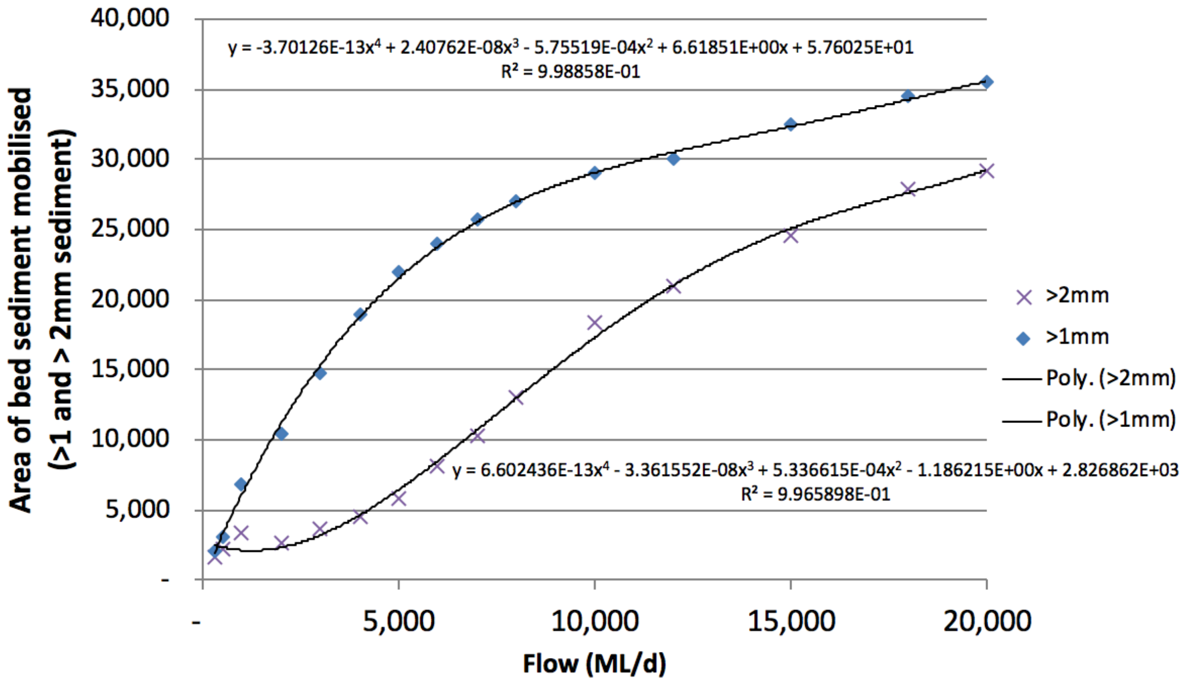


Figure 3-17. Bed mobilisation. Area of bed sediment mobilised for gravels (crosses) and medium-grained sands (blue solid) at McCoy's Bridge.

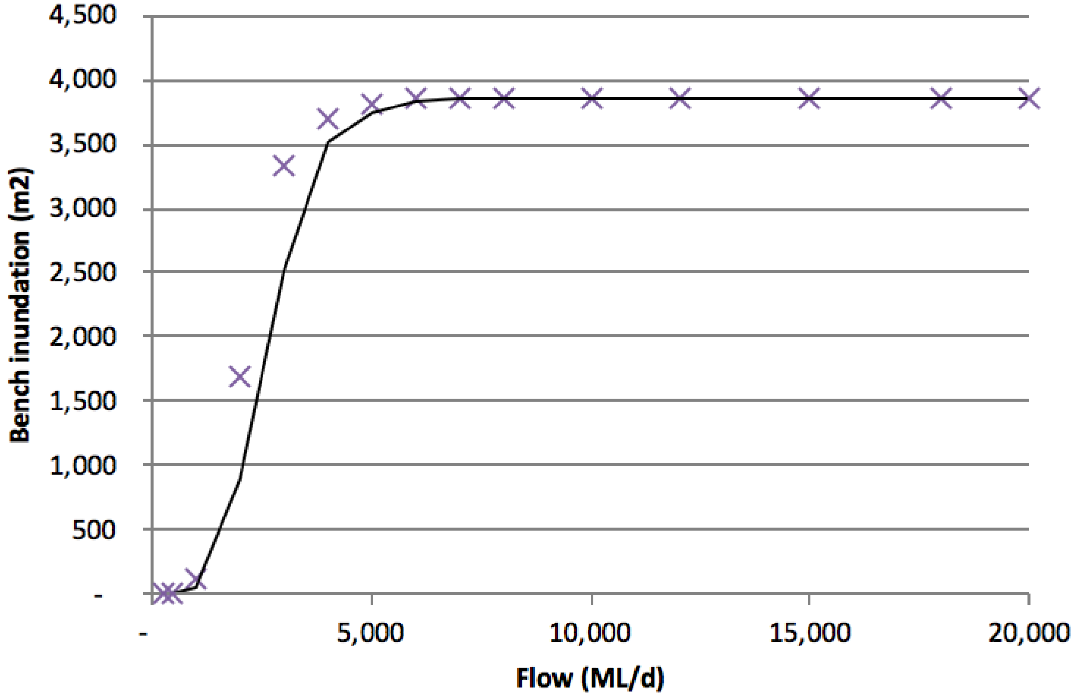


Figure 3-18. The area of bench inundation for McCoy's Bridge.

3.6 Bank condition

3.6.1 Bank Condition Methods

Equipment used for this monitoring program consists of 200 erosion pins (50 pins at each of the four sites), which are 300 mm long bicycle spokes with colour coded heat shrink (Figure 3-19, left). Each pin is inserted into the bank so that 25 mm is exposed. Erosion pins are located at five different elevations (up to approximately bankfull) on each of ten transects at each site. Changes in surface level relative to each erosion pin are made using digital callipers (see Figure 3-19, right). Qualitative assessments are also made at each transect on erosion process, failure mechanism, and weakening process (see proforma in the SOP; Webb et al. 2018).

Figure 3-19. (left) Colour coded erosion pins inserted at each transect to indicate location/elevation on the river bank and measured by digital callipers, and (right) field placement.

Recordings with positive values (relative to starting position) indicate bank retreat (erosion) and negative values indicate bank aggradation (deposition). Data presented in this report are from the program start (January 2015) to April 2018. Further details on the erosion assessment protocol can be found in (Vietz et al. 2018).

3.6.2 Hydrologic variables and statistical analysis

Flow metrics that have been used as model predictors to characterise effects on bank condition of environmental flows include (Table 3-5):

- Inundation duration
- Peak flow magnitude
- ADWP – maximum dry weather period prior to inundation
- Flow volume
- Rate of draw down – average, maximum and minimum

Table 3-5. Flow metrics used for comparison with bank erosion measurements.

Flow metric	Description	Justification
Duration of inundation	How many days an erosion pin is under water between surveys	The time over which a bank is exposed to inundation and/or flowing water influences bank wetting and saturation, and the effect of cumulative shear stress on erosion. Similarly, deposition may be a function of cumulative time over which sediments can move through the water column to deposit on the bank.
Peak flow magnitude	Peak flow of an event that inundated an erosion pin between surveys (the maximum if multiple peaks are experienced)	Erosion/deposition may be driven by the maximum shear stress associated with an event, with sediment bank sediments being mobilised, or accumulated (if scoured from elsewhere) during the period around peak flows.
Flow volume	Volume of flow of the event above the level of the pin that inundates an erosion pin	A metric that combines duration and magnitude to assess the 'work' being done on the bank by water.
Maximum dry weather period	Maximum number of days without inundation of the pin prior to inundation	Banks may become more sensitive to erosion when inundated if they are allowed to dry out completely, inducing desiccation and cracking of clay-rich sediment particles.
Maximum dry weather period by season	Maximum number of days without inundation of the pin prior to inundation by 'hot season' (Nov-Apr) and 'cold season' (May-Oct)	Banks may become more sensitive to erosion when inundated if they are allowed to dry out completely, inducing desiccation and cracking of clay-rich sediment particles. This is hypothesised to be more severe during the hot season when banks can rapidly dry.
Average and maximum rate of drawdown	Day 2 discharge divided by Day 1 discharge for the falling limb of a flow event	The rate at which flow recession from an event occurs can impact on bank erosion through surcharging a bank (saturating) and affecting the support provided by the water while the bank is saturated. If the rate of recession is too great mass failure (slumping) can occur, particularly on steep banks.

A hierarchical Bayesian logistic regression model was used to identify the relationship between the flow metrics and bank erosion/deposition. The probability of erosion and deposition was assessed as a function of each metric, as experienced by the erosion pin based on 13 measurements. Other flow characteristics, such as bank notching (a horizontal demarcation in the bank associated with the water level surface), have been considered based on observations but have not been assessed statistically.

3.6.3 Bank Condition and Flows Statistical Model

The occurrence of erosion or deposition (*y*) for pin *j* at site *k* during season *s* and survey *i* is a Bernoulli-distributed event with probability *p*. This is driven by a global average erosion/deposition across all sites in the absence of inundation (*int*), plus the effect of the inundation metric being analysed (*eff.I*) for each site/season combination, multiplied by the metric value for that survey (*I*). There is a random effect of site (*eff.site*) that acknowledges that local conditions may enhance or retard overall erosion/deposition, a random effect of survey (*eff.surv*) to capture any seasonal or other systematic differences among survey periods in erosion/deposition, and a random effect of pin (*eff.pin*) to account for the repeated measures taken for each pin.

The key update in the 2017–18 model is that the effect of cold/hot (wet/dry) seasons are included in the inundation metric effects (*eff.I*), besides the existing differences across sites. As a result, effect of inundation is drawn from individual distribution for each site/season combination (Equation 3a).

$$y_{ijk} \sim Bern\left(p_{ijks}\right) \qquad \text{Equation 1}$$

$$logit(p_{ijks}) = int + eff.inund_{ks} \times inund_{ijks} + eff.site_k + eff.survey_i + eff.pin_{jk} \qquad \text{Equation 2}$$

$$eff.inund_{ks} \sim Normal(\mu\_inund_{ks}, \sigma\_inund) \qquad \text{Equation 3a}$$

$$eff.site_k \sim Normal(0, \sigma\_site) \qquad \text{Equation 3b}$$

$$eff.survey_i \sim Normal(0, \sigma\_survey) \qquad \text{Equation 3c}$$

$$eff.pin_{jk} \sim Normal(0, \sigma\_pin) \qquad \text{Equation 3d}$$

3.7 Results Bank Condition

3.7.1 Bank activity and the effect of season

Bank erosion and deposition exhibited similar responses to flow during the previous periods. Erosion and deposition are still highly variable both in time and space, but both still occur relatively commonly even at the same location. In general erosion is still slightly greater than deposition or no change, for Loch Garry, McCoys Bridge and Yambuna (Figure 3-20). Darcy’s Track, however, experiences less change with ‘no change’ more dominant. Significant erosion (considered as > 30 mm) is still only a small fraction of the measurements (3- 7%).

In general, deposition occurs more often in the cold season, while erosion occurs more often in the hot season (Figure 3-20). There is no distinct difference in significant erosion across seasons.

The most notable difference in flow conditions during this hot period was associated with the larger IVT flows delivered between January and May 2018. Higher rates of bank activity were identified following this period, both erosion and deposition, but the proportional change from the previous three-year period is small.

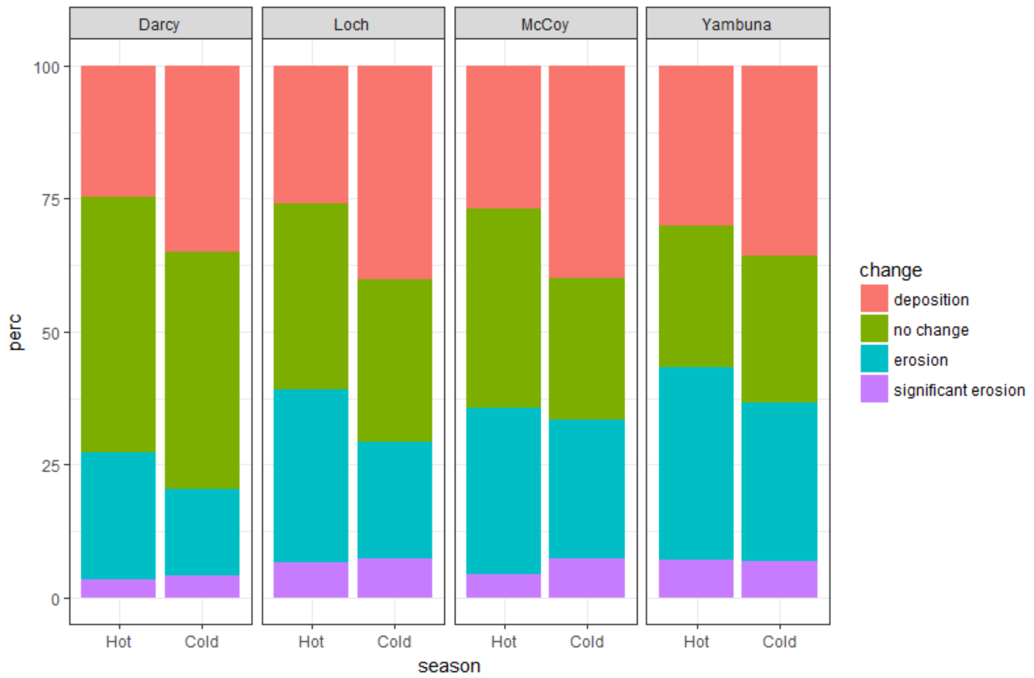


Figure 3-20. Proportion of deposition, no change, erosion and significant erosion in measurements over the past four years, separated by season (hot: summer/autumn, cold: winter/spring) and site. Sample sizes are: 816, 754, 740 and 801 for Darcy's Track, Loch Garry, McCoy's Bridge and Yambuna, respectively.

3.7.2 Flow impacts on bank activity

- In general, inundation period (especially the hot season) is the strongest predictor for probabilities of both erosion and significant erosion (Table 3-6).
- For erosion, all sites except for Yambuna have slightly higher flow effects during summer, but this is not statistically significant at the 95% level. For significant erosion, there is no clear difference in the flow effects of seasons at all sites (Table 3-6).
- For probabilities of deposition, both inundation and peak flow magnitudes are strong predictors (Figure 3-21 and Table 3-6).
- For deposition, Darcy's Track and McCoy's Bridge have slightly lower flow effects during summer, but this is not statistically significant at the 95% level (Figure 3-20).
- Average, maximum and minimum rate of draw down have similar effects on bank activities (similar predictive capabilities), so only the strongest predictor, maximum rate of draw down, is presented (Table 3-6). The effect of drawdown is only significant for the McCoys Bridge site.
- Assessing the effect of environmental flows (counterfactual simulations), there is a slightly higher probability of bank activities due to environmental flow, across all sites (Figure 3-22). These increases are more distinct during winter than summer.

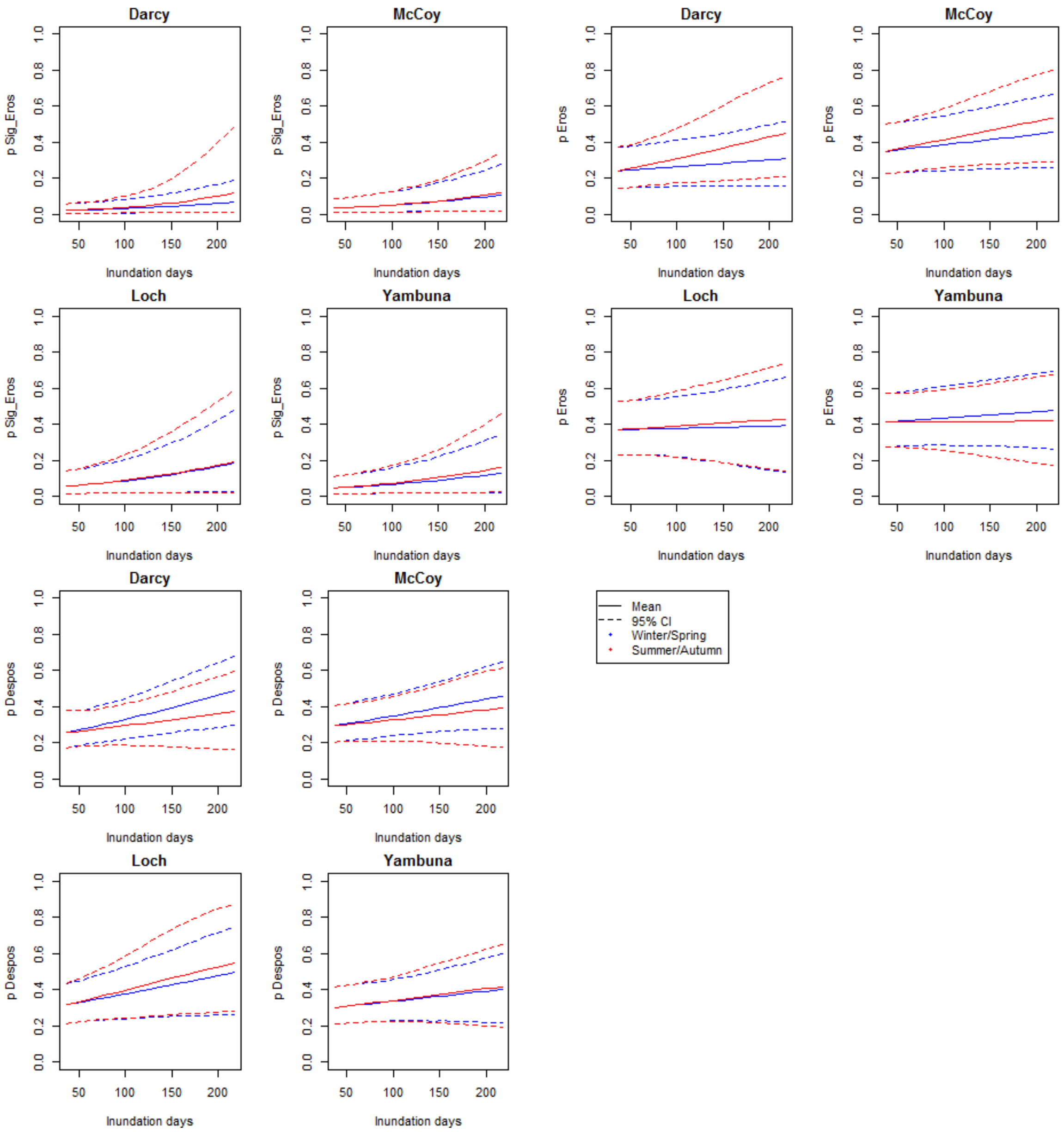


Figure 3-21. Probability of erosion of > 30 mm (a), > 0 mm (b) and deposition < 0 mm (c), with increases in the duration of inundation. For each erosion level, results are shown four for sites (Darcy's Track, Loch Garry, McCoy's Bridge and Yambuna) in individual panels. The solid line is the median probability of erosion with the dotted lines encompassing the 95% credible interval for the estimate.

Table 3-6. 95% credible intervals of regression coefficients (Eff.I) for three erosion levels and for each flow metric. Bold values represent instances where there is a relationship between erosion/deposition and flow metric.

Bank activity	Predictor/ season		Darcy		Loch		McCoy		Yambuna	
			2.50%	97.50%	2.50%	97.50%	2.50%	97.50%	2.50%	97.50%
Significant erosion (>30)	Inundation	Cold	-0.008	0.516	-0.017	0.617	0.031	0.531	-0.040	0.494
		Hot	0.071	0.933	-0.070	0.692	-0.001	0.621	0.004	0.664
	Peak	Cold	-0.007	0.685	0.054	0.742	-0.013	0.518	-0.040	0.646
		Hot	-0.554	0.507	-0.552	0.507	-0.140	0.874	-0.247	0.820
	ADWP	Cold	-0.366	0.545	-0.212	0.550	-0.286	0.525	-0.129	0.692
		Hot	-0.484	0.457	-0.453	0.416	-0.188	0.479	-0.316	0.420
	Volume	Cold	-0.545	0.222	-0.787	0.397	-3.799	0.258	-0.648	0.182
		Hot	-1.593	0.627	-1.683	0.620	-1.292	0.011	-0.993	1.360
	Max RDD	Cold	-0.874	0.392	-0.882	0.207	-1.926	-0.026	-0.520	0.552
		Hot	-0.430	0.434	-0.415	0.346	-0.728	0.104	-0.215	0.536
Erosion (>0)	Inundation	Cold	-0.074	0.227	-0.291	0.226	-0.045	0.263	-0.114	0.224
		Hot	0.005	0.551	-0.260	0.334	0.001	0.460	-0.261	0.203
	Peak	Cold	-0.084	0.269	-0.196	0.207	-0.037	0.291	-0.054	0.330
		Hot	-0.322	0.231	-0.564	0.164	-0.176	0.394	-0.239	0.368
	ADWP	Cold	-0.228	0.308	-0.425	0.092	-0.161	0.357	-0.158	0.357
		Hot	-0.538	-0.019	-0.355	0.114	-0.259	0.102	-0.077	0.329
	Volume	Cold	-0.141	0.180	-0.391	0.167	-0.566	0.214	-0.169	0.216
		Hot	-0.323	0.352	-0.609	0.227	-0.189	0.172	-0.403	0.390
	Max RDD	Cold	-0.107	0.532	-0.162	0.340	-0.058	0.562	-0.351	0.179
		Hot	-0.179	0.208	-0.147	0.258	-0.406	0.012	-0.084	0.275
Deposition (<0)	Inundation	Cold	0.063	0.436	-0.030	0.429	0.014	0.338	-0.086	0.280
		Hot	-0.117	0.345	0.013	0.618	-0.142	0.318	-0.129	0.334
	Peak	Cold	0.312	0.746	0.149	0.619	-0.009	0.357	0.122	0.565
		Hot	0.137	0.774	0.337	1.198	0.110	0.836	-0.060	0.734
	ADWP	Cold	-0.208	0.202	-0.066	0.320	-0.209	0.190	-0.091	0.287
		Hot	-0.027	0.350	-0.210	0.182	-0.101	0.202	-0.135	0.203
	Volume	Cold	-0.081	0.315	-0.407	0.343	-2.276	0.122	-0.281	0.214
		Hot	-0.516	1.219	-0.352	2.121	-0.739	-0.132	-1.355	0.460
	Max RDD	Cold	-0.538	-0.004	-0.289	0.224	-0.865	-0.116	-0.310	0.199
		Hot	-0.310	0.059	-0.363	0.028	-0.344	0.024	-0.235	0.165

3.7.3 Counterfactual effects without Eflow

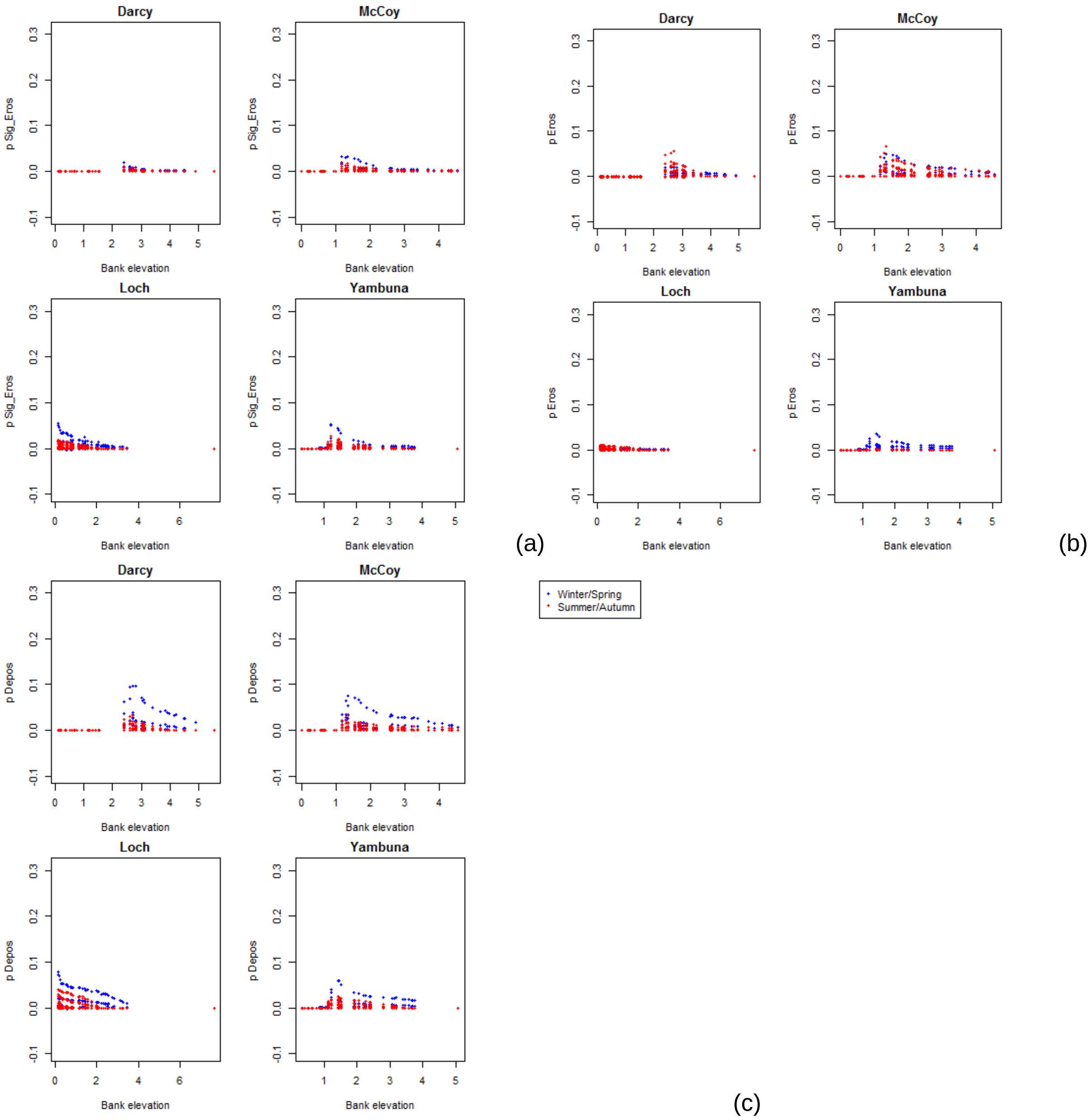


Figure 3-22. Without environmental flows: Effect of the environmental flow component on the probability of significant erosion (> 30 mm, a), erosion (> 0 mm, b) and deposition (< 0 mm, c), at each erosion pin, relative to bank elevation (m).

3.8 Discussion

The erosion evident during the summer period points to the important role of riverbank drying in erosion processes. Since peak magnitude and total flow volume were not significantly related to riverbank erosion it can be inferred that the dominant erosion mechanism is not related to high velocities but the influence of inundation on the bank. This supports the role of sub-aerial preparation of bank sediments whereby drying of clay-rich soils (desiccation) leads to cracking and preparation of banks for erosion during subsequent inundation (Figure 3-23).

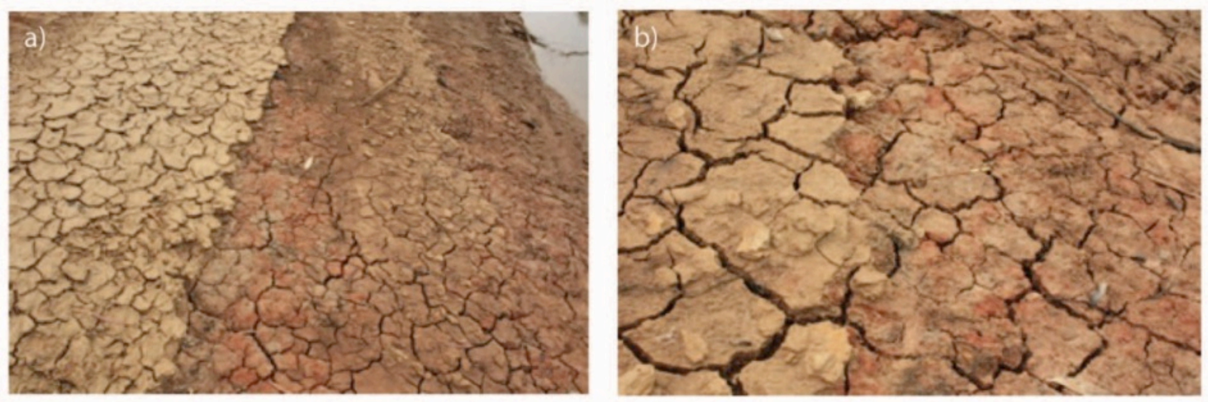


Figure 3-23. a) Drying of clay-rich sediments prepares bank materials for removal during subsequent inundation, and b) note erosion pin exposed (centre picture) at the Yambuna site with 5 cm of erosion measured following a fresh as desiccated sediment was removed.

The role of bank erosion relative to bank vegetation has yet to be linked. Zones of deposition did provide niches for vegetation colonisation. Anecdotally, vegetation plays an important role in the resistance of banks to erosion. Sub-aerial preparation of banks as a result of drying and cracking is exacerbated when vegetation is not available to shade soils. In addition, root wads enhance structural integrity. Deposition is also enhanced by vegetation through increased roughness, encouraging further vegetation establishment. Data from bank condition and riparian vegetation assessments will be synthesised to understand relationships in the year 5 annual report.

The main challenges encountered with the bank condition monitoring have been with accessing erosion pins when prolonged water levels are encountered. This was most problematic during high winter/spring flows in 2016 (and to a lesser extent in 2017) and IVT flows in Summer/Autumn 2018. This challenge has not unduly impacted on the results and can be addressed, in part, by close collaboration between the operators and field staff undertaking the assessments.

4. Stream Metabolism

4.1 Introduction

Whole stream metabolism measures the production and consumption of dissolved oxygen gas (DO) by the key ecological processes of photosynthesis and respiration (Odum 1956). Healthy aquatic ecosystems need both processes to generate new biomass (which becomes food for organisms higher up the food chain) and to break down plant and animal detritus to recycle nutrients to enable growth to occur. Hence metabolism assesses the energy base underpinning aquatic foodwebs. The relationships between these processes are shown in Figure 4-1.

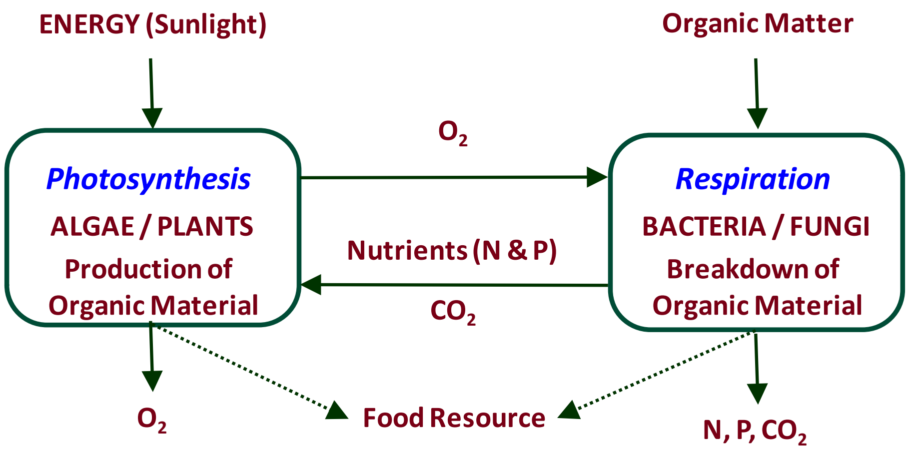


Figure 4-1. Relationships between photosynthesis, respiration, organic matter, dissolved gases and nutrients.

Metabolism is expressed as the increase (photosynthesis) or decrease (respiration) of DO concentration over a given time frame; most commonly expressed as (change in) milligrams of dissolved oxygen per Litre per day (mg O_2/L/Day). Typical rates of primary production and ecosystem respiration range over two orders of magnitude, from around 0.2 to 20 mg O_2/L/Day with most measurements falling between 0.5 and 10 mg O_2/L/Day.

If process rates are too low, this will limit the amount of food resources (bacteria, algae and water plants) for consumers. This limitation will then constrain populations of larger organisms including fish and amphibians. Rates *are* expected to vary on a seasonal basis as warmer temperatures and more direct, and longer hours of, sunlight contribute to enhancing primary production. Warmer temperatures and a supply of organic carbon usually result in higher rates of ecosystem respiration (Roberts and Mulholland 2007).

In general, there is concern when process rates are too high. Greatly elevated primary production rates usually equate to algal bloom conditions (or excessive growth of plants, including duckweed and *Azolla*), which may block sunlight penetration, killing other submerged plants, produce algal toxins and large diel DO swings - overnight, elevated respiration rates can drive the DO to the point of anoxia (no dissolved oxygen in the water). Such conditions have been observed in several sites in the Goulburn River in previous years of the LTIM project. When an algal bloom collapses, the large biomass of labile organic material is respired, often resulting in extended anoxia. Very low (or no) DO in the water can result in fish kills and unpleasant odors. Bloom collapse often coincides with release of algal toxins; hence the water becomes unusable for stock and domestic purposes as well.

Sustainable rates of primary production will primarily depend on the characteristics of the aquatic ecosystem. Streams with naturally higher concentrations of nutrients (e.g. arising from the geology), especially those with very open canopies (hence lots of sunlight access to the water) will have much higher natural rates of primary production than forested streams, where rates might be extremely low due to heavy shading and low nutrient concentrations. Habitat availability, climate and many other factors also influence food web structure and function. Uehlinger (2000) demonstrated that freshes with sufficient stream power to cause scouring can 'reset' primary production to very low rates which are then maintained until biomass of primary producers is re-established.

Some but not all of the organic carbon created through gross primary production is respired within the first 24 hours. Such respiration is performed by the autotrophs (primary producers) themselves and closely associated heterotrophic communities. Although there is a large amount of variability in the proportion respired 'immediately' Hall and Beaulieu (2013) estimate that on average 44% of new organic carbon created is respired before it can move into higher trophic levels.

4.2 Area specific evaluation questions

To determine the contribution of Commonwealth environmental water in selected areas, and to improve understanding of the relationship between specific watering actions and ecological objectives for assets, the following questions are being addressed. This information also forms the basis of Basin-scale evaluation – where area-level results are scaled up to the Basin level.

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
How does the timing and magnitude of CEW delivery affect rates of Gross Primary Productivity and Ecosystem Respiration in the lower Goulburn River?	Yes	Apart from the initial dilution effect (as seen in previous years), there was no consistent effect of flow increases (including those from CEW delivery) across the 4 sites on rates of either GPP or ER over the period of record when metabolism is expressed as mg O_2/L/day. However, there was a positive effect of flow increases, even those constrained within channel, on total amounts of GPP and ER expressed as mass (load) of organic carbon per day. Bayesian modelling found no evidence for lag effects (increased metabolic rates from 1-15 days after the onset of the event) when metabolism was expressed as mg O_2/L/day.	Based on regression of daily discharge versus rates of GPP and ER, and on calculated loads of organic carbon. Flow was categorized according to Section 4.2.2. There was sufficient variability of flow levels to detect any significant positive regressions. Bayesian models relating daily estimates of GPP and ER to water velocity were used to determine optimal lag periods for both GPP and ER.
How do stream metabolism responses to CEW in the lower Goulburn River differ from CEW responses in the Edward Wakool system where the likelihood of overbank flows is	Partially	As found in previous years, stream metabolism rates were slightly lower in the Goulburn River compared to the Edward-Wakool. The actual CEW and natural flows in the Edward-Wakool, notably the absence of any overbank flows precluded assessment of the second part of the question. Both systems had very low bioavailable nutrient concentrations (especially phosphorus) which was a significant	Based on daily estimates of rates of GPP and ER regressed with daily flow rate, photosynthetically active radiation (PAR) (GPP only), and temperature. Monthly nutrient sampling was assumed to be representative of nutrient concentrations most/all of the time.

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
higher and nutrient concentrations are generally much lower?		constraint on GPP (and affected ER too). Very low bioavailable phosphorus (and nitrogen) is the reason metabolic parameters are at the lower end of international values. Thus similarities between these two systems are much stronger than the differences, despite the difference in typical discharges,	

4.3 Main findings from the stream metabolism monitoring program

4.3.1 Findings from 2017–18

The main findings from 2017–18 monitoring can be summarised as:

- Contrary to the prevailing thought at the start of this project that water needed to reach backwaters, flood-runners and even the floodplain before any positive outcome would be seen in metabolism, by considering the amount of organic carbon created by GPP (and consumed by ER), this report shows that even small increases in discharge that remain within channel can have positive benefits for the energy (‘food’) underpinning aquatic foodwebs.

- Categorization of flows into ‘bands’ allowed the pooling of metabolism data, thereby averaging out variation due to season and daily weather conditions and hence provided an excellent way of comparing metabolism in different flow regimes.

- Using McCoy’s Bridge as an example, increasing discharge from a median very low flow of 312 ML/day to a median low flow of 960 ML/day will result in an increase of 73% in organic carbon produced by GPP and an increase in 19% in the amount of organic carbon respired.

- It is still suggested that larger flow increases that do move the water out of channel and then back again will provide even greater benefit due to the introduction of higher organic carbon and bioavailable nutrient concentrations.

- DO concentrations in 2017–18, as in 2015–16 and 2016–17, dropped to very low levels that raise concerns about the immediate effects on aquatic biota. However, unlike 2016–17, anoxia did not occur this year. The origin of the low DO regime is clearly water entering the Goulburn River from the tributaries downstream from Goulburn Weir as the Day Road site was unaffected. These poor water quality events were of moderate duration (typically 1-2 weeks before DO levels reverted to ‘normal’) and appeared to be stochastic, arising from intense summer storms in the northern half of the Goulburn Catchment.

- Statistical modelling between discharge (or water velocity) and metabolic parameters found no evidence of a lag effect in response to flow increases.

4.3.2 How these build on findings from years 1 to 3

These findings build on findings from years 1 to 3 by:

- The major initial effect of flow increases on stream metabolism is a rapid decrease in rates of GPP and ER (in mg O_2/L/day) through simple dilution.

- The results again demonstrated the potential for very low DO (if not anoxic) conditions that can quickly develop with intense summer rainfall events in the northern half of the catchment.

4.4 Methods

The stream metabolism and water quality measurements were performed in accordance with the LTIM Standard Operating Procedure (Webb et al. 2018).

Water temperature and dissolved oxygen were logged every ten minutes with one ZebraTech DO logger placed in each of the four sites in zones 1 (Day Rd[1], Darcy's Track) and 2 (McCoy's Bridge, Loch Garry). Data were downloaded and loggers calibrated approximately once per month depending on access. In some months, downloads were delayed by high water levels preventing access to the loggers (too far underwater). Light (PAR) loggers were also deployed in open fields at Shepparton and Nagambie (Tahbilk); these data were downloaded every few months.

In accord with the LTIM Standard Protocol, water quality parameters (temperature (^oC), electrical conductivity (mS/cm), dissolved oxygen (%), pH, and turbidity (NTU)) were also measured as spot recordings at two sites within each river reach during deployment and maintenance of the DO loggers.

Water samples were collected from the same two sites within each zone used for the metabolism measurements, to measure:

- Total Organic Carbon (TOC)

- Dissolved Organic Carbon (DOC) and Particulate Organic Carbon (POC)

- Nutrients (Ammonia (NH_4^+), filtered reactive phosphorus (FRP), dissolved nitrate + nitrite (NOx), Total Nitrogen (TN) and Total Phosphorus (TP))

In accord with the LTIM Standard Protocol, water quality parameters (temperature (^oC), electrical conductivity (μS/cm), dissolved oxygen (%), pH, and turbidity (NTU)) were measured fortnightly.

The stream metabolism measurements were performed in accordance with the LTIM Standard Operating Procedure (Hale et al. 2014). After discussions at the annual LTIM forum in Sydney in July 2016, it was decided that an updated version of the BASE model (BASEv2) would be used for analysing the 2015-16 metabolism data and all data sets from that time onwards. This change was a result of the paper published by Song et al. (2016) which showed that our BASE model could be improved by changing from stepwise progression and fitting using each data point to integrated (whole data set) fitting and progression using modelled data.

Water temperature and dissolved oxygen for stream metabolism determination were logged every ten minutes with at least one logger placed at each of the four study sites; at sites 1, 3 and 4, loggers were placed at the upstream and downstream end of these sites. Data were downloaded and loggers calibrated approximately once per month, and more frequently (often fortnightly) during summer time to avoid problems found in previous years with probe biofouling. Downloading also depended upon access, as described below. Light and depth loggers were also deployed and data were downloaded on an approximately monthly basis. The data collected by the loggers were used to calculate daily

[1] The site at Day Rd was chosen in 2015-16 to replace the Moss Rd site used in 2014-15 and has been used since that time. It was found that the Moss Rd site was simply too close to the weir wall and almost no usable data (met acceptance criteria) were obtained.

average temperature and dissolved oxygen concentrations for each of the sites from June 2017 (to complete a full winter data set encompassing July and August 2017) to mid-June 2018.

Acceptance criteria for inclusion of daily results from the BASEv2 model (Grace et al. 2015) in the data analysis presented here were established at the July 2015 LTIM Workshop in Sydney and adjusted at the corresponding meeting in July 2016. These criteria were: the fitted model for a day must have an r^2 value of at least 0.90 *and* a coefficient of variation for GPP, ER and K parameters of < 50%.Finally the convergence measure, PPP, must lie between 0.1 and 0.9. Outside of this range means inadequate convergence a strong likelihood that the model parameters do not provide a robust fit to the data (an implausible model). Finally, to exclude occasional data days that meet all these requirements but produce unrealistically high (or) low estimates of GPP and ER, the reaeration coefficient, K, was constrained to the range 0.1 < K, 15 / day. These very infrequent parameter excursions occur due to the high correlation between ER and K. A K value < 0.1 / day is extremely unlikely as this would be a lower reaeration than from a completely undisturbed still water surface; values > 15 /d ay indicate highly turbulent flow (which is common in small streams but very unusual in low gradient larger rivers such as the Goulburn.

The evaluation of the combined four-year data set required all of the data used for stream metabolism to be rerun on the BASEv2 program to ensure a common methodology across time (years). Changes to the optimization routine during 2017 has meant that there are now many more days that meet the acceptance criteria for inclusion in the analysis presented here. It is important to note however, that there has been no change at all in the fundamental model explaining how dissolved oxygen changes as a function of time due to primary production, respiration and reaeration (See the Stream Metabolism Foundation Report; Grace 2015) for further details).

4.4.1 Derived Stream Metabolism Metrics

Up until this point, GPP and ER have been expressed solely in the units from the original measurements, namely mg O_2/L/Day. Two new complementary units will be explored in this report:

The amount (mass) of organic carbon created/consumed each day in a one km stream reach (kg orc C/km/day). This unit is intended to relate to the amount of organic carbon required by the food web in that stream reach each day and eventually to the sustainable stocking capacity for native fish in that reach on the assumption that this capacity is resource (food) limited. If there is insufficient organic carbon (which equates to ‘energy supply’) being provided at the base of the food web, then higher trophic levels, including fish, will be resource limited, and irrespective of improvements to habitat, availability of flows to trigger spawning etc., native fish populations will remain constrained. There is much to be done in the future to quantitatively establish this link between primary production and the energetic needs of fish, but this metric provides the basis for such links to be made. The unit is calculated by simply multiplying the daily metabolism estimate by the cross sectional area of water in the channel that day at the gauging station. The cross-sectional relationships were provided by Guarino and Stewardson (2018) and are listed as Table 4-1. Finally, conversion to organic carbon involves a factor of 12/32 (ratio of atomic mass of C and molecular mass of O_2). This factor does not include any physiological efficiency factor for converting oxygen to organic carbon which typically is in the range 0.8-1. Given the exploratory use of this metric, concern over conversion efficiency at this stage is unwarranted.

The mass of oxygen (or organic carbon, see above) produced per day. This is calculated by multiplying the GPP or ER in mg O_2/L/day by the number of Litres discharged that day. As has been noted in two previous Basin Level Evaluation Reports (Grace 2016, 2017), the most notable effect of discharge on metabolism is an immediate reduction due to the dilution effect of the additional water. However the fact there is now more water may mean that the overall amount of oxygen (hence organic carbon) produced or consumed that day may actually increase.

A third derived unit 'Areal metabolism units (g O_2/m^2/Day)' is being explored as part of the Basin Level Evaluation. This unit expresses GPP and ER as oxygen produced/consumed per m^2 of stream (or sediment) surface per day. It is obtained simply by multiplying the original units by mean water depth in the reach. Most metabolism reports worldwide use this areal approach although the estimation of mean reach depth is fraught given the challenge of measuring the actual depth at a sufficient number of transects and estimating the reach length integrated by the dissolved oxygen probe (approximated by 3v/K, where v is the mean water velocity in m/Day and K is the reaeration coefficient; Reichert et al., 2009).

4.4.2 Flow 'Categories'

As part of the ongoing development of hydrological descriptors of flow regimes undertaken in LTIM, discharge can be grouped according to the flow stages developed by Stewardson and Guarino (2018) and reproduced from their report as Figure 4-2 here:

According to Stewardson and Guarino (2018), the various flow levels are established as:

- Very low flows: flows less than the lowest flow in the unimpacted monthly flow series or 2% of mean unimpacted flow, whichever is greater.
- Medium low flows: flows that fall below the 95th percentile exceedance flow in the unimpacted monthly flow series or 10% of the mean unimpacted flow, whichever is greater.
- Low freshes: flow spells that raise water levels at least 1/8th of the height of the bank above the medium low flow level.
- Medium freshes: flow spells that raise water levels at least 1/4 of the height of the bank above the medium low flow level
- High freshes flow spells that raise water levels at least 1/2 of the height of the bank above the medium low flow level.

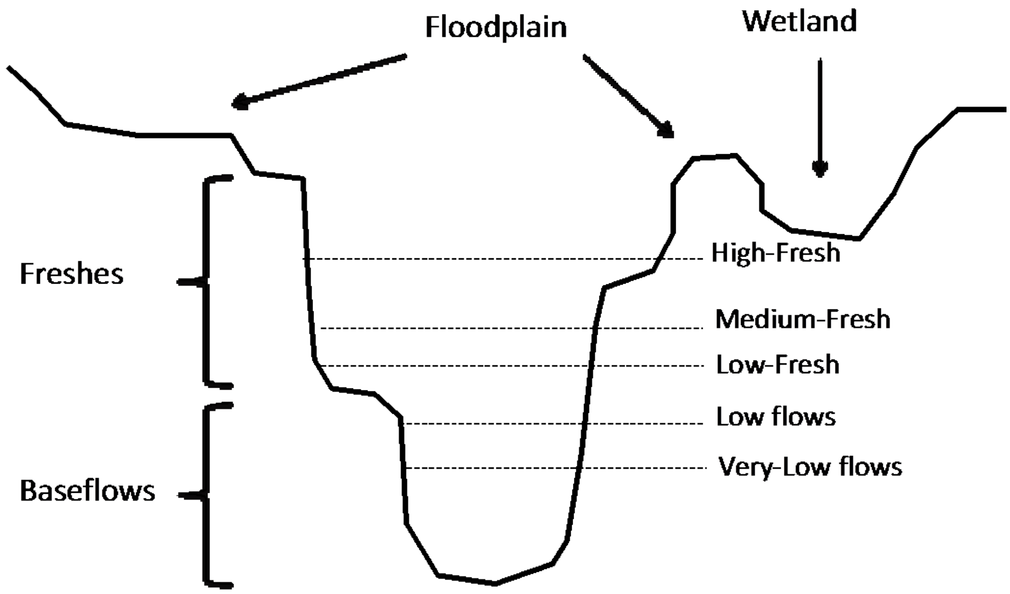


Figure 4-2. Flow stages according to Stewardson and Guarino (2018).

The flow thresholds associated with these stages was provide by (Stewardson and Guarino 2018) – the data relevant to the Goulburn River metabolism sites are presented in Table 4-1.

Table 4-1. Flow Thresholds (ML/Day) for Goulburn River stream metabolism monitoring sites

Site Name	LTIM Site	Modelled Natural Flow Site Name	Very Low	Moderate Low	Low Fresh	Medium Fresh	High Fresh	Finalised Bankfull
Murchison	Moss/Day Rd	405253 – Goulburn @D/S Goulburn Weir	252	868	4224	12600	12600	57168
McCoy's	McCoy's Bridge	405232 – Goulburn @D/S McCoy's Bridge	312	960	4157	11714	11714	50278
Shepparton	Darcy's Track	405272 – Goulburn @U/S Shepparton	253	910	3862	10774	10774	45772

4.4.3 Statistical Modelling

Relationships between discharge and gross primary production (GPP), ecosystem respiration (ER) and net primary production (GPP – ER = NPP) were analysed using a hierarchical Bayesian linear regression of the metabolism endpoint against discharge (log transformed) and temperature. First-order auto-regressive terms in the model tested for (and compensated for) the lack of temporal independence in the daily data.

$$y_{ij} \sim Normal(\mu_{ij}, \sigma) \qquad \text{Equation 1}$$

$$\mu_{ij} = int_j + eff.Q_j \times \log(Q_{ij}) + eff.Te_j \times Te_{ij} + eff.Light_j \times Light_{ij} + ac.e^{-eff.d(d_{ij}-d_{i-1,j})}(y_{i-1,j} - (int_j + eff.Q_j \cdot \log(Q_{i-1,j}) + eff.Te_j \cdot Te_{i-1,j}))$$

Equation 2

Stream metabolism (GPP, ER and NEP), represented by *y*) on day *i* and at site *j*, is distributed normally around a mean metabolism of μ and standard deviation of σ. Mean metabolism on day *i* and at site *j* is a linear function of log of discharge indicator (*Q*), temperature (*Te*) and light (*Light*). The intercept (*int*), and the effect of discharge indicator (*eff.Q*), effect of temperature (*eff.Te*) and effect of light (*eff.Light*) are specific for each site (temperature and light are both included because use each only one of them lead to non-convergence of the model). *int,eff.Q* and *eff.Te* were modelled hierarchically. All prior distributions were minimally informative.

The *ac* term quantifies the extent to which a data point can be estimated from the point preceding it (i.e., autocorrelation). This term is multiplied by a weighted exponential function parameterized by the term *eff.d*, which is the extent to which autocorrelation breaks down with increasing temporal separation of data points ($d_i - d_{i-1}$).

We have explored the following model predictands (y_{ij} in Equation 1):

- GPP
- ER
- NEP

For the GPP, the following discharge indicators have been tried as model predictors (Q_j in Equation 2):

- Discharge

- Velocity
- Delta discharge – difference between discharge and previous-day discharge

For each candidate predictor, the model has also been run for scenarios that assumed a lag of between 0 and 15 days, where the lag represents the time between discharge on a day and a resulting effect on metabolism (e.g. time needed for algal populations to increase after an influx of nutrients on a particular day). The optimal lag was determined as the lag at which the R^2 of the model is at its maximum, which indicates the best model fit.

4.5 Results

In this report, results are presented and analysed over two time frames: the 2017–18 sampling year and the entire four year period of record. Many data in this report are presented as boxplots. Box (or Box & Whisker) plots provide a convenient and simple visual means of comparing the spread of data. The boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. “Whiskers” above and below the box indicate the 90th and 10th percentiles. Values beyond this, called “far outside values” or “outliers” are plotted as single circles.

Estimates of Gross Primary Production and Ecosystem Respiration for the 4 sites were produced using the BASEv2 model (Grace et al. 2015), updated according to (Song et al. 2016). The periods of data logger deployments are listed in Table 4-2 along with the number of days’ data that meet the extended acceptance criteria ($r^2 > 0.90$, coefficient of variation for all of GPP, ER and K < 50%, 0.1<PPP<0.9). The % compliance data for 2014–15, 2015–16 and 2016–17 are included for comparison (Day Rd was not a site for 2014–15).

There were periods during the year 4 logger deployment when various loggers were out of the water for short periods, submerged too deeply by high flow to safely retrieve resulting on some data loss through flat logger batteries and other minor issues. This information is recorded in the Appendix to this chapter (Section 4.7).

Table 4-2. DO Logger Deployment and Data Acceptance Information, 2017-18.

Site	First Date	Last Date	Number of Days with data	Compliant Days using BASEv2	2017-18% of total days in compliance	2016-17% of total days in compliance	2015-16 % of total days in compliance	2014-15 % of total days in compliance
Day Road	7/9/17	13/6/18	240	111	46	54	27	n/a
Darcy’s Track	7/9/17	13/6/18	155	80	52	53	28	72
Loch Garry	8/9/17	13/6/18	218	101	46	51	33	38
McCoy’s Bridge	1/6/2017	13/6/18	345	278	81	56	48	66

As discussed later, a month long period of low dissolved oxygen occurred from around 2/12/17 and affected all sites except Day Road.

4.5.1 Water Temperature and Dissolved Oxygen

Figure 4-3 displays the mean daily water temperature and mean daily dissolved oxygen concentrations, collected from the DO loggers, at all four sites over the entire deployment period.

Gaps in the data reflect logger maintenance, and logger inaccessibility resulting in battery failure as detailed in Annex A.

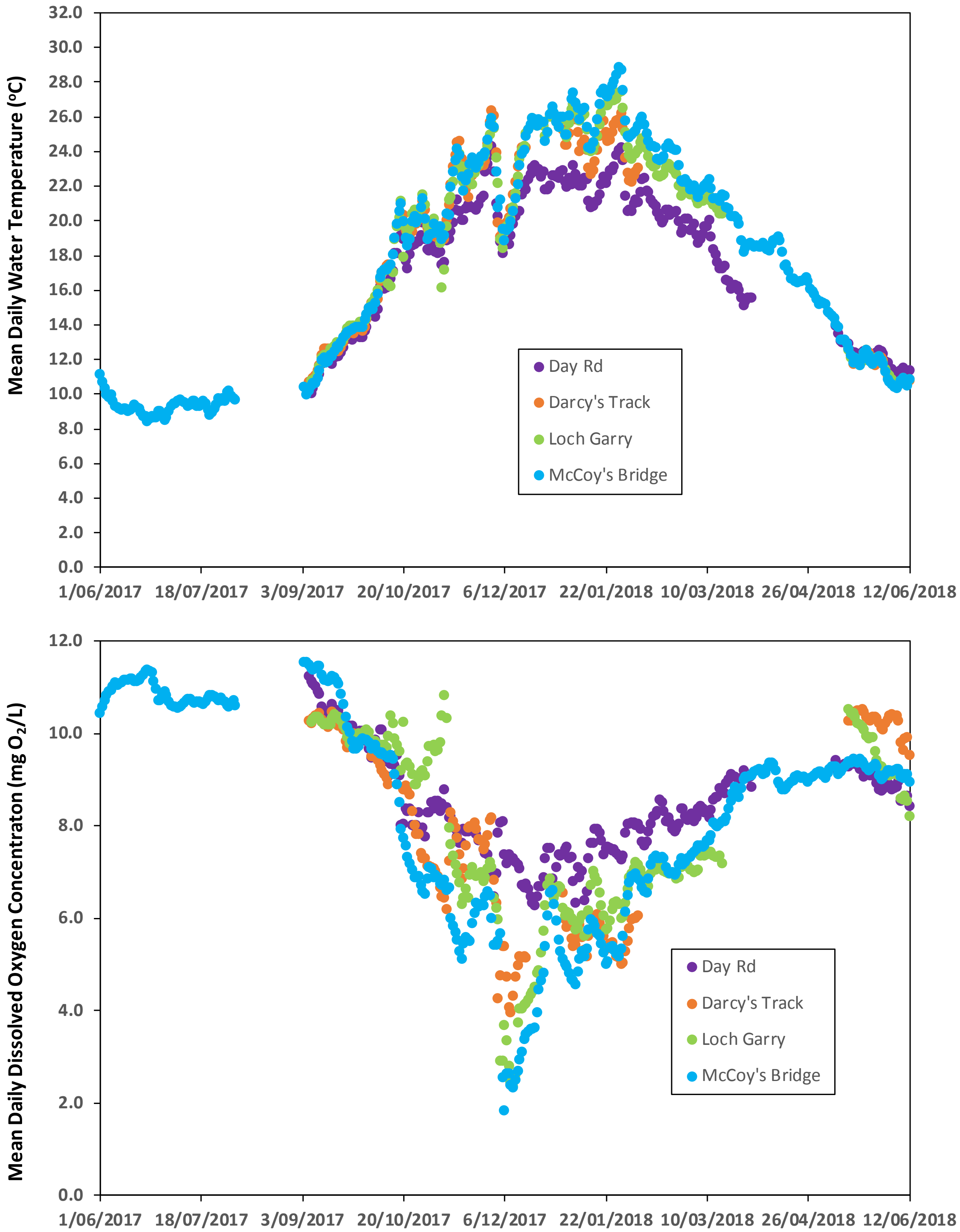


Figure 4-3. Mean Daily Water Temperature and Dissolved Oxygen Concentration for the four study sites 2017-18.

The temperature profiles shown in Figure 4-3 conform to expected behaviour with the warmest average daily temperatures occurring in mid-late summer. Similarly the general pattern of decline in dissolved oxygen concentrations in the warmer months is also expected due to the decreasing solubility of oxygen gas in water as the temperature of that water increases. The water temperature is noticeably lower at Day Rd and this is most likely the result of the site being relatively close to the outflow from Goulburn Weir. It is an underflow weir hence bottom water is released from the Nagambie Lakes which will be cooler than the surface water, especially during daytime in summer when solar irradiance (and hence epilimnetic heating) is at a maximum. This temperature difference between Day Rd and the sites further downstream can be several degrees. This temperature differential is partially overcome by Darcy's Track but does emphasize the generic finding that 'cold water pollution' can extend for large distances downstream of weir structures. The effect is fairly minimal here but definitely identifiable.

One very noticeable feature in Figure 4-3 is the sharp drop in temperature of around 8^{o}C (much less at Day Rd) occurring in early December 2017. There is a corresponding sudden decrease in dissolved oxygen at the three more downstream sites at the same time. To assist visualization of the pattern of behaviour, Figure 4-4 presents the percentage DO saturation data (combining temperature and dissolved oxygen concentration) over the six-week period starting November 21, 2018.

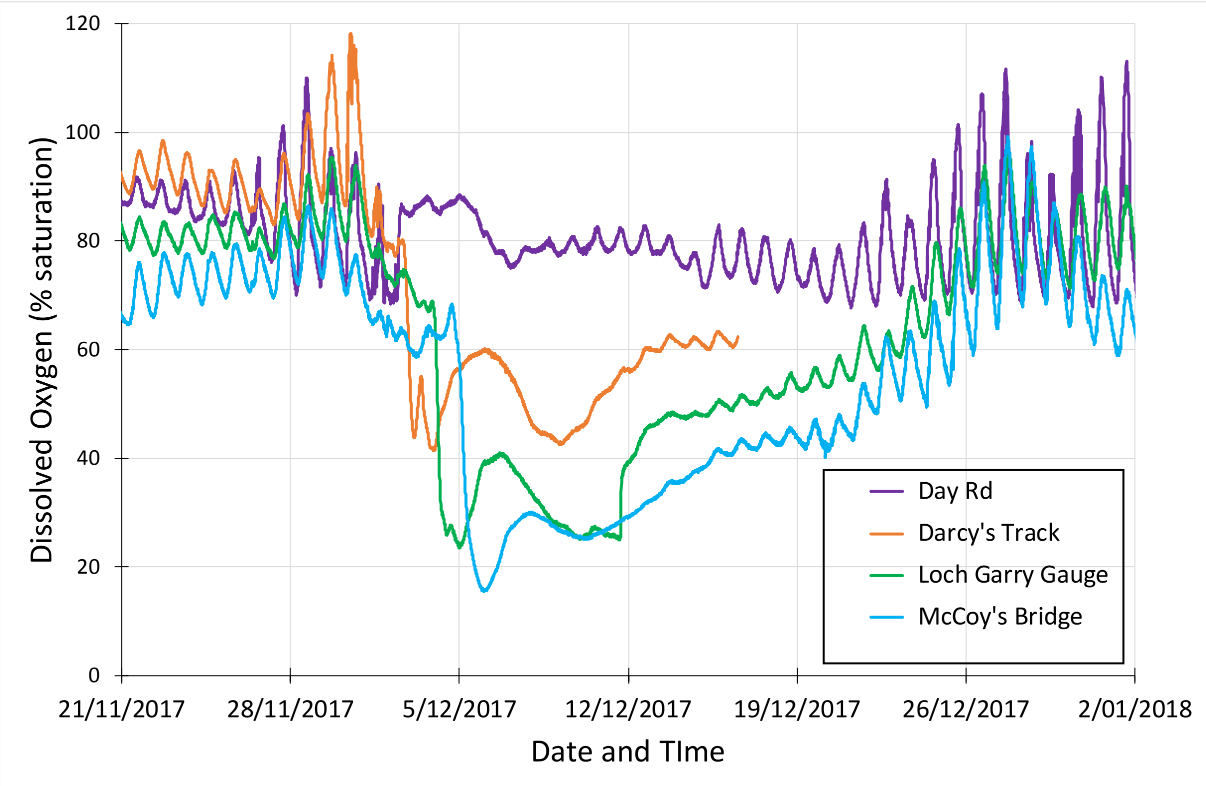


Figure 4-4. Percentage Dissolved Oxygen Concentration for the four study sites from November 21, 2017 through to January 2, 2018.

There was a very large, intense rain event in parts of the Goulburn-Broken catchment over the period December 1-3, 2017, which delivered up to 200 mm of rain (GBCMA 2017). High inflows from several tributaries of the Goulburn downstream from Day Road (Seven, Pranjip and Castle creeks) contributed water with very low dissolved oxygen. The offset in time for the rapid %DO decrease shown in Figure 4-4 is directly related to the water travel time between these sites. The influence of this event on oxygen concentrations lasted until the end of December. Although % Dissolved Oxygen dropped to 20% saturation at McCoy's Bridge, which is certainly low enough to cause oxygen stress in fish, at no stage did anoxic conditions develop. Thus the fish kills associated with anoxia seen in this river reach in the previous two years did not eventuate. This recurring pattern of a very heavy rain event during summer causing very low DO or even anoxic conditions appears to be a regular feature of this reach. In this case, a small amount of additional water was released from Goulburn Weir (1500 ML/Day) over several days to improve water quality.

4.5.2 Metabolic Parameters

From the results of modelling using BASEv2, the parameter estimates for GPP, ER, the reaeration coefficient K and the ratio of Gross Primary Production to Ecosystem Respiration ratio (P / R) for all 4 sites monitored, derived from all days meeting the acceptance criteria, are presented in Table 4-3.

Table 4-3. Summary of primary production (GPP) and ecosystem respiration (ER) rates, P/R ratios and reaeration coefficients for the four study sites, 2017-18.

Parameter	Day Rd (n =111)			Darcy's Track (n =80)		
	Median	Min	Max	Median	Min	Max
GPP (mg O_2/L/Day)	2.20	0.39	22.9	1.23	0.06	4.04
ER (mg O_2/L/Day)	9.26	0.80	40.7	2.28	0.19	10.7
P / R	0.26	0.07	0.96	0.51	0.02	1.14
K (/Day)	7.06	0.19	14.6	1.18	0.13	4.61
Parameter	Loch Garry (n =101)			McCoy's Bridge (n =278)		
	Median	Min	Max	Median	Min	Max
GPP (mg O_2/L/Day)	0.98	0.10	2.95	0.96	0.03	4.87
ER (mg O_2/L/Day)	3.16	0.21	22.20	2.36	0.06	12.3
P / R	0.36	0.01	2.26	0.38	0.01	9.87
K (/Day)	1.31	0.11	9.22	1.39	0.16	4.41

Each metabolic parameter in Table 4-3 is expressed as a median with minimum and maximum values also included. The median provides a more representative estimate without the bias in the mean arising from a relatively few much higher values. The median GPP values from all four sites fall within a very narrow range of 0.96 (McCoy's Bridge) to 2.20 (Day Road) mg O_2/L/Day. The range of median ER values is equally small, varying from 2.28 mg O_2/L/Day at Darcy's Track up to 3.16 mg O_2/L/Day at the Loch Garry Gauge. The one exception to this is the much larger ER for Day Road (9.26 mg O_2/L/Day). The reason for this much higher respiration rate at Day Road is likely to be from relatively labile organic matter exported from the Goulburn Weir. There does not appear to be any longitudinal trend in results for either GPP or ER, but this conclusion remains speculative given the significant periods with no data at different sites. e.g. missing two months of data from March-May 2018 for Day Road and Loch Garry due to high water levels preventing logger access resulting in battery failure.

Figure 4-5 to Figure 4-8 to Fig. 2-8 display the daily rates of GPP, ER and then P/R ratio at all 4 sites. The daily flow data are also plotted in each figure. The P/R ratio indicates the relative importance of oxygen production to oxygen consumption within a river reach on a particular day. As GPP can vary significantly depending on the daily weather, looking at this ratio over only a short period can give misleading results. A ratio of > 1 indicates that more oxygen (and hence organic carbon) is being produced than is being consumed.

The P/R ratios (medians 0.26 to 0.51) are lower than those observed in previous years. This again is attributed to the inclusion of winter-time metabolism data in 2017–18. GPP rates are constrained much more by season than ER rates. The median values indicate that, in general and on a daily basis, significantly more oxygen is consumed in these reaches than is produced. However, the maximum P/R ratios indicate that at times, oxygen production is as high Day Road, Darcy's Tack) or much higher (Loch Garry, McCoy's Bridge) in comparison to consumption via ecosystem respiration. In most cases, as observed in previous years, high P/R is typically due to lower ER rates than massively increased GPP.

To put these metabolic rates into a global context, a summary of world-wide stream metabolism data (mostly from the USA) shows that GPP and ER values are typically in the range 2-20 mg O_2/L/day (Bernot et al. 2010, Marcarelli et al. 2011) based on an assumption that the average water depth of 1 m (to convert the areal units of many reports to the volumetric units used in LTIM). Hence these Goulburn River data fall towards the bottom end of this global range. Whether these low rates, mirrored across the southern Basin, reflect a system under stress or are indicative of 'normal' rates for Australian lowland rivers should become more apparent as LTIM evolves. Publication of a significantly more extensive data set (from the USGS) covering many more biomes in the USA is imminent and will show that the Basin metabolic rates are *not* unusually low.

It is interesting to compare the metabolic data for 2017–18 with that found for previous years. Note that all data presented in Table 4-4 below has been calculated using the BASEv2 model, and with the current acceptance criteria; hence comparisons are not confounded by use of different models.

Table 4-4. Comparison across four years of median primary production (GPP) and ecosystem respiration (ER) rates, P/R ratios and reaeration coefficients for the four study sites.

Site	Day Rd				Darcy's Track			
Year	**14-15**	**15-16**	**16-17**	**17-18**	**14-15**	**15-16**	**16-17**	**17-18**
n	n/a	39	78	111	109	43	52	80
GPP (mg O_2/L/Day)		1.10	1.82	2.20	1.53	1.41	2.25	1.23
ER (mg O_2/L/Day)		2.08	4.55	9.26	1.34	2.76	3.87	2.28
P/R		0.93	0.48	0.26	1.00	0.70	0.58	0.51
K (/Day)		3.38	6.79	7.06	1.45	2.08	2.02	1.18

Site	Loch Garry				McCoy's Bridge			
Year	**14-15**	**15-16**	**16-17**	**17-18**	**14-15**	**15-16**	**16-17**	**17-18**
n	52	47	70	101	193	92	114	278
GPP (mg O_2/L/Day)	1.36	2.10	1.76	0.98	1.39	1.67	1.46	0.96
ER (mg O_2/L/Day)	1.24	2.78	1.96	3.16	1.03	1.76	2.89	2.36
P/R	1.07	0.90	0.73	0.36	1.15	0.68	0.58	0.38
K (/Day)	2.11	1.87	1.7	1.31	3.02	1.97	1.53	1.39

The data shown in Table 4-4 highlight different behaviours between Day Road and the three other sites further downstream. For those three sites, GPP in 2017–18 was lower than previous years for all sites other than Day Road where the highest median GPP was recorded. The lower median GPP is explained by the inclusion of more winter data (for McCoy's Bridge) and first-time winter data for the other two sites. Rates are naturally lower due to cooler temperatures and shorter days with less intense sunshine during winter, reducing the overall median. Seasonal comparisons will be made in the Year 5 report. The observed slight increase in median GPP for Day Road in 2017–18 compared to previous years also incorporates winter data in 2017–18. The origin of this increase is not obvious, as bioavailable nutrient concentrations are very low (next section) and comparable with the other three sites. It is worth highlighting that the median ER rate at Day Road is much higher in 2017–18 (9.26 mg O_2/L/Day) than previously at this site or any other site in this Selected Area over the duration of the LTIM Project. Higher rates at Day Road strongly suggest significant effects from the Goulburn Weir upstream although the exact nature of this enhanced metabolism remains unclear at this stage.

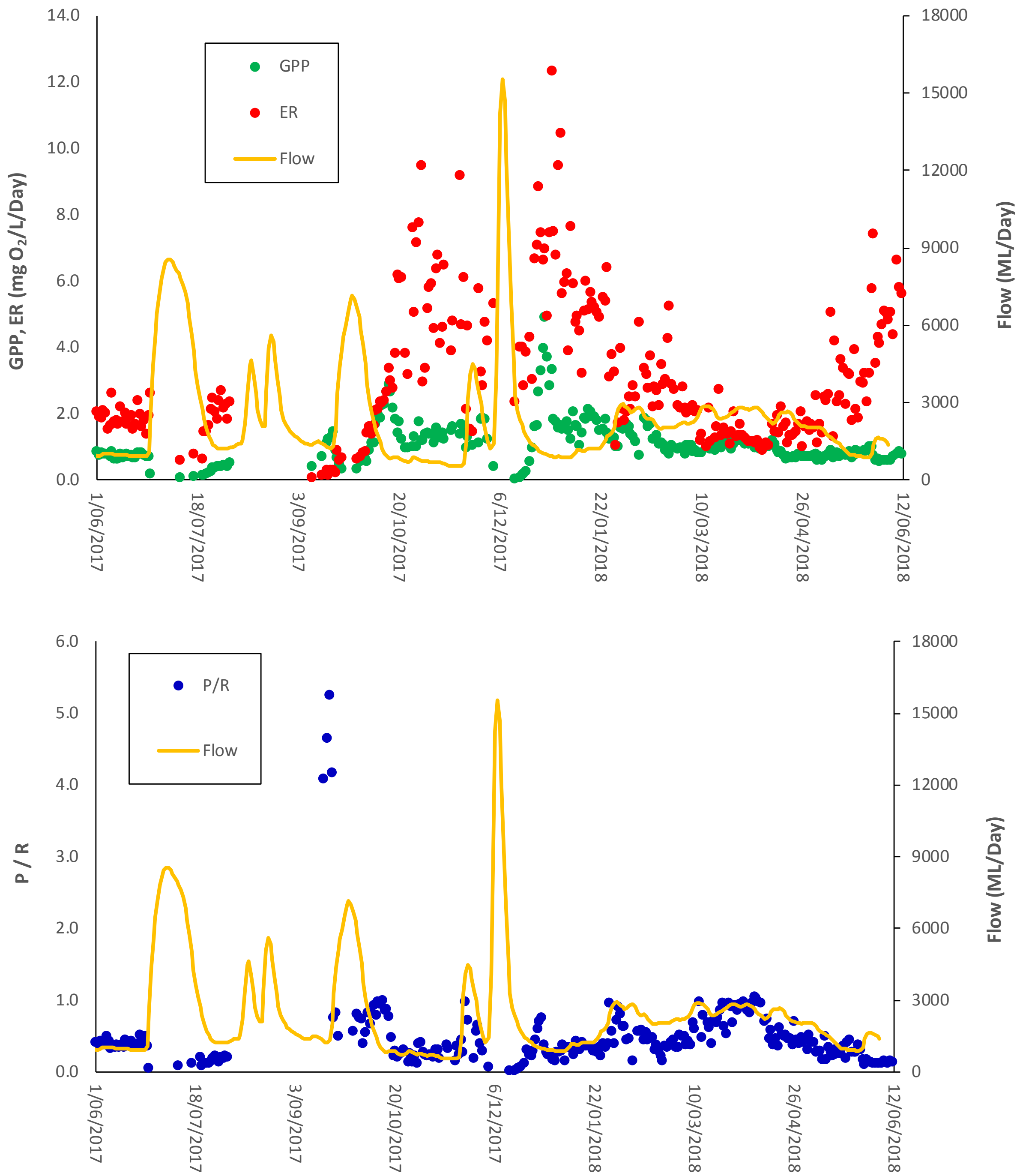
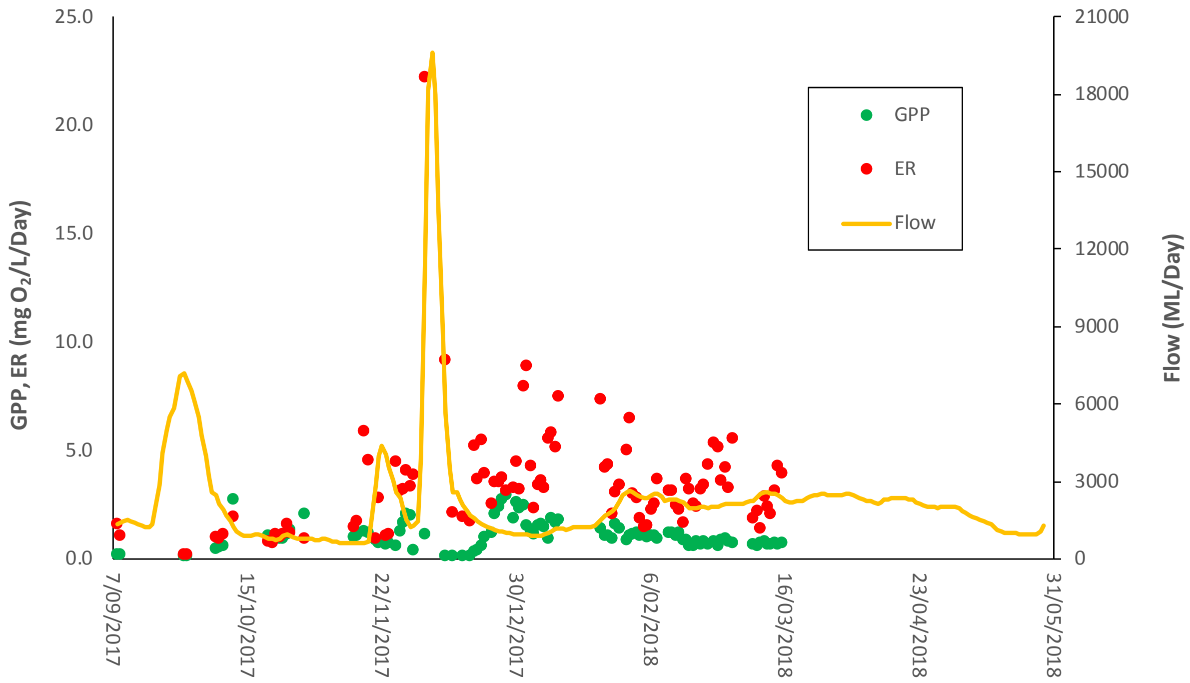


Figure 4-5. Stream Metabolism-Flow Relationships for McCoy's Bridge (Zone 2) from June 2017 to June 2018: a) Gross Primary Production and Ecosystem Respiration; b) P / R ratio



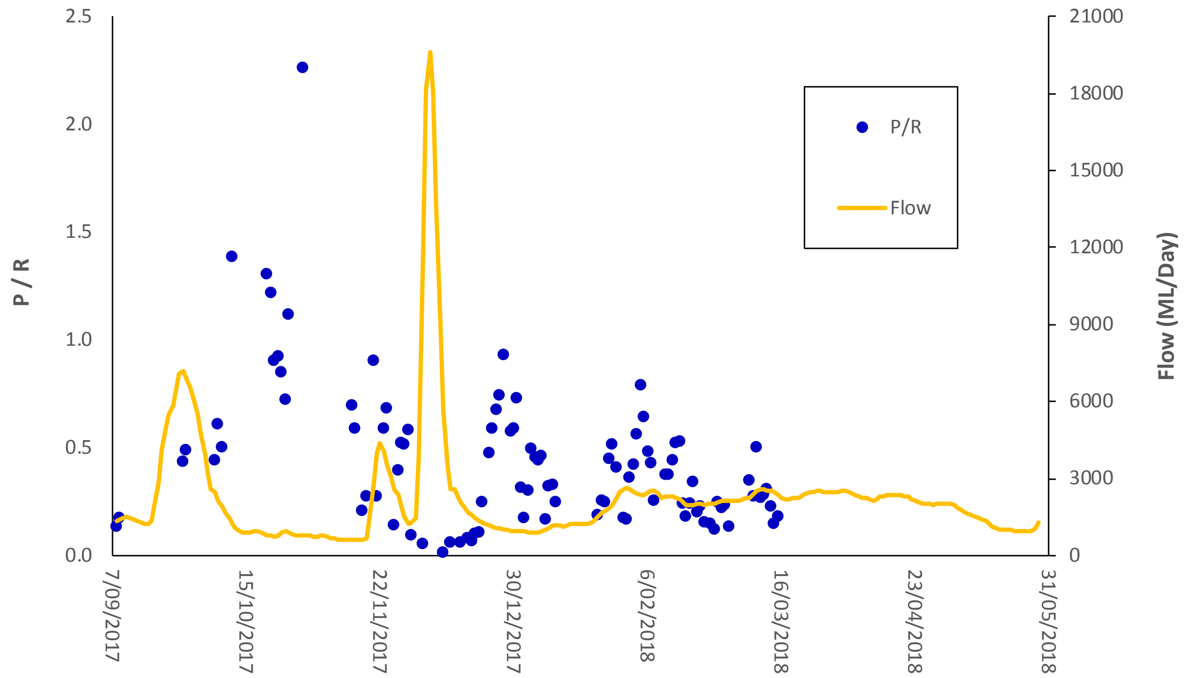


Figure 4-6. Stream Metabolism-Flow Relationships for Loch Garry (Zone 2) from August 2015 to April 2016: a) Gross Primary Production and Ecosystem Respiration; b) P / R ratio.

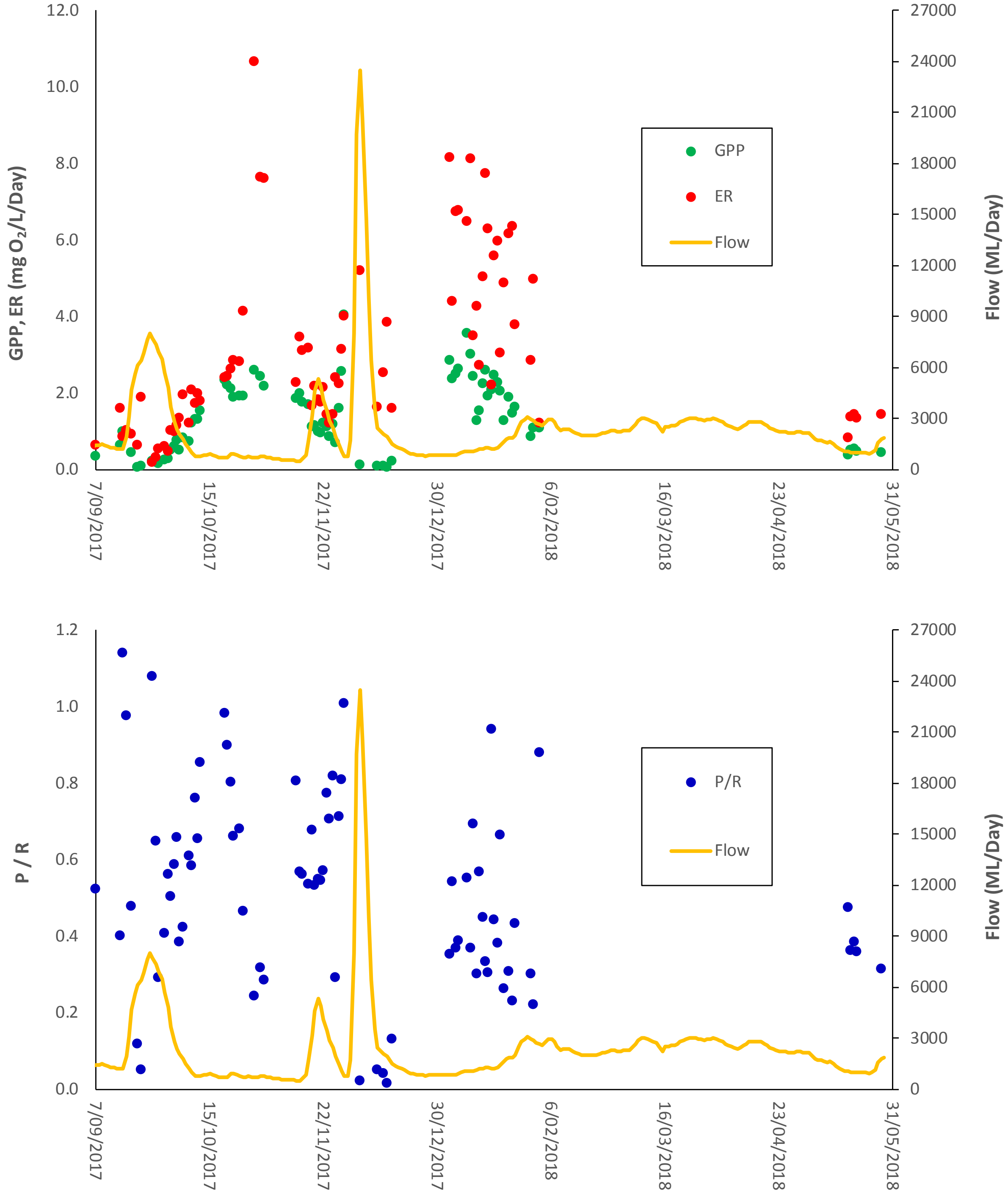


Figure 4-7. Stream Metabolism-Flow Relationships for Darcy's Track (Zone 1) from September 2017 to June 2018: a) Gross Primary Production and Ecosystem Respiration; b) P / R ratio.

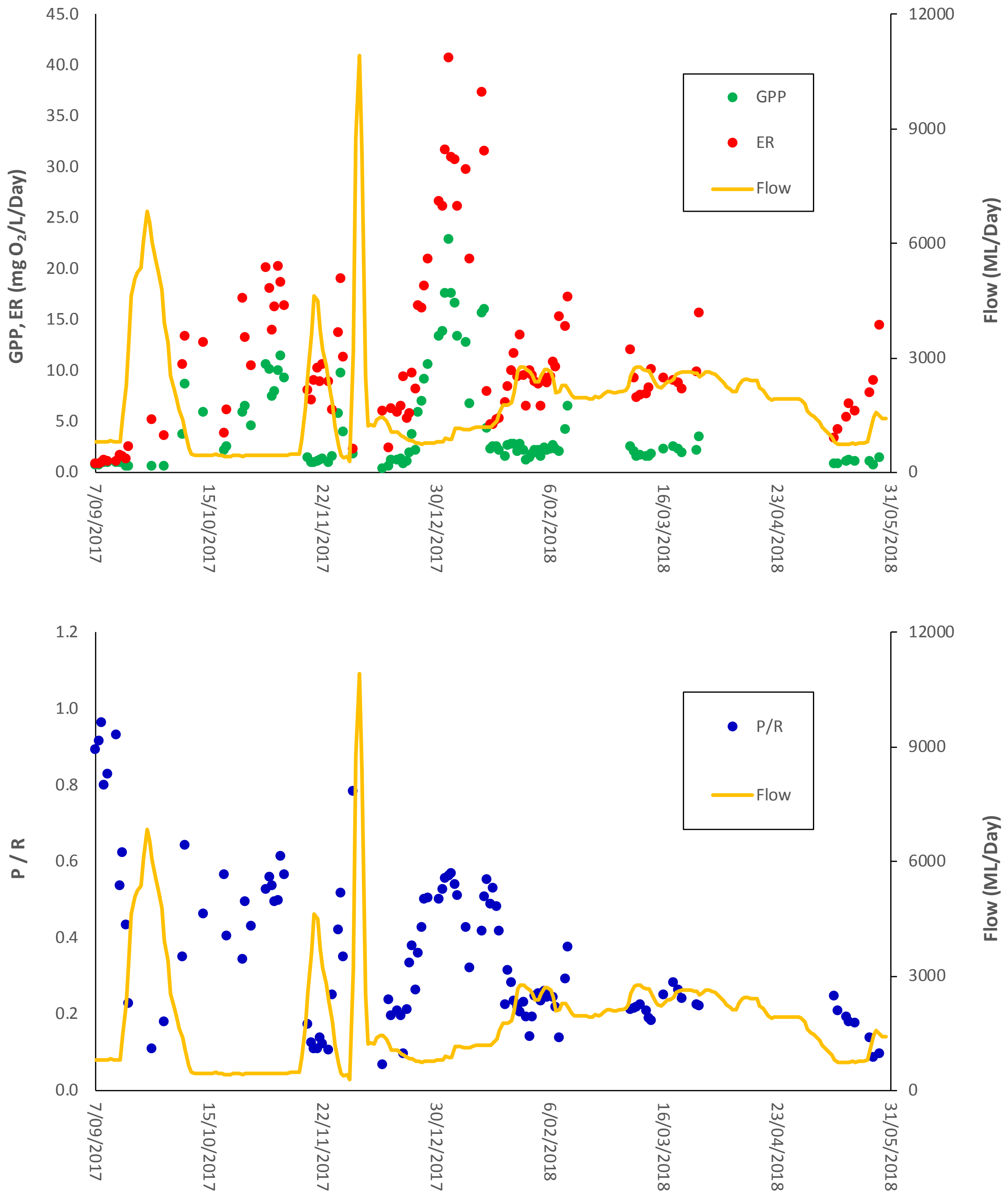


Figure 4-8. Stream Metabolism-Flow Relationships for Day Road (Zone 1) from September 2017 to June 2018: a) Gross Primary Production and Ecosystem Respiration; b) P / R ratio.

In comparing results from Table 4-4 and Table 4-5, it is important to note that Goulburn results make up around 21% of the overall database used to generate Table 4-5. Nevertheless, the range in median GPP over the four Goulburn sites is similar to the overall LTIM result. Very little winter data were collected in any of the six Selected Areas over the first three years of the LTIM Project, hence it is not surprising that the Year 4 results in Table 4-3 are also lower than the pooled LTIM values. With the obvious exception of the Day Road median ER rate for 2017–18 highlighted above, all of the ER rates in Table 4-4 are very similar to those across the six Selected Areas. It is highly likely that the same factors constraining primary production (mainly nutrients) and ecosystem respiration (organic carbon supply) are important in the entire southern Basin as well as specifically the Goulburn River (It is likely that GPP in the northern Basin is constrained by light availability rather than nutrient supply).

Table 4-5. Summary LTIM Stream Metabolism Statistics for all six Selected Areas, 2014–17.

	n	Median	Mean	Std Dev	Std Error	25th %ile	75th %ile
GPP (mg O2/L/Day)	4807	1.65	2.19	2.2	0.03	1.06	2.64
ER (mg O2/L/Day)	4807	2.72	3.67	3.4	0.05	1.47	4.62

4.5.3 Investigating the Basal Drivers for Metabolism

As noted in previous annual reports, primary production is expected to depend upon temperature and light (PAR) while respiration is also expected to increase with increasing temperature. Consequently, linear regressions were performed between the two metabolic parameters and the anticipated explanatory variables. The results of these regressions are presented in Table 4-6.

Table 4-6. Exploration of Linear Relationships between the metabolic parameters (GPP and ER) and, Light and Temperature for the four study sites, 2017-18. Statistical significance was inferred at $p < 0.05$.

Site		GPP vs Temp	GPP vs Light	ER vs Temp	Light vs Temp
Loch Garry	r^2	0.18	0.14	0.068	0.093
	P	< 0.001	< 0.001	0.009	0.002
	slope	0.08	0.13	0.20	0.16
McCoy's Bridge	r^2	0.32	0.3	0.17	0.54
	P	< 0.001	< 0.001	< 0.001	< 0.001
	slope	0.059	0.13	0.14	0.32
Darcy's Track	r^2	0.38	0.16	0.33	0.34
	P	< 0.001	< 0.001	< 0.001	< 0.001
	slope	0.12	0.17	0.27	0.27
Day's Rd	r^2	0.092	0.24	0.17	0.19
	P	0.001	< 0.001	< 0.001	< 0.001
	slope	0.35	1.05	0.83	0.21

As expected, both GPP and ER daily rates were positively correlated with mean daily water temperature (Table 4-6) for all sites. There was a large degree of variability (scatter) in these regression plots (an example is shown below for McCoy's Bridge GPP vs Average Daily Water Temperature as Figure 4-9), partially due to the effects of discharge and light (for GPP). GPP was positively correlated with light at each site. Unsurprisingly, plots of Light versus Water Temperature were strongly positively correlated. Solar irradiance provides both light and heat to the water surface, so days of higher and more intense sunshine result in warmer water temperatures. This finding does mean that subsequent data analysis must take into account this covariance. This issue is explored further in the Bayesian modelling described in Section 4.5.4.

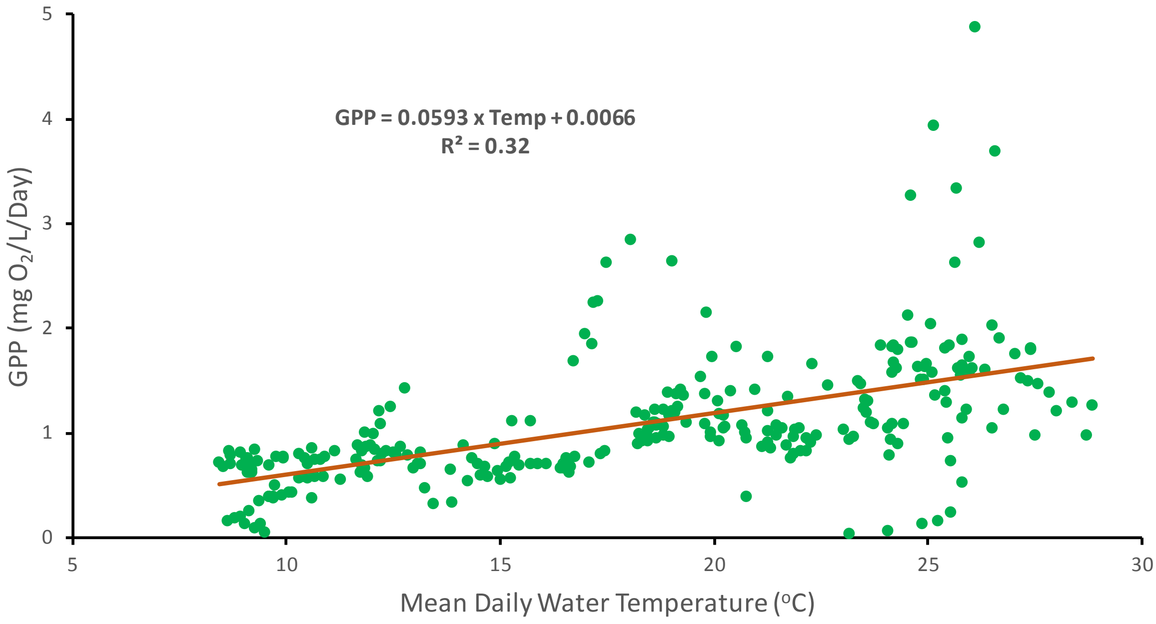


Figure 4-9. The Relationship between Daily Gross Primary Production and Average Daily Water Temperature at the McCoy's Bridge site, June 2017 to June 2018 (n = 278).

As sampling progressed from spring into summer, GPP rates generally increased due to a combination of longer days (more sunlight) and warmer temperatures. Rates peaked in the weeks after the large flow in early December (See Figure 4-5 for example). Rates then slowly declined into autumn and then winter, although the absence of many data in the March – May period due to battery failure makes it impossible to follow the GPP decline through autumn. A key point is that although the GPP rates varied with time (season) and location, the magnitude of the variability was very small. Rates were constrained within a narrow range (Table 4-3).

Nutrient concentrations from the four sites were determined on the samples that were collected approximately monthly during the DO probe deployment, downloading and maintenance. These data are presented in Table 4-7. Also included in the table are data from the long term monitoring program at McCoy's Bridge (DELWP 2015). Dating back to 1990, data were collected weekly up until December 2013, when monthly sampling was instituted.

The key finding from Table 4-7, is that, consistent with the three previous years, the concentrations of bioavailable nutrients in the Goulburn River at all 4 sites were very low. In particular, the bioavailable phosphorus concentration FRP, was consistently below 0.01 mg P/L with a couple of exceptions at McCoy's Bridge. It is very difficult to draw any conclusions about the effects of flow events (including Commonwealth environmental water) on nutrient concentrations as monitoring does not occur over the changing hydrograph; instead it is performed when the DO loggers are downloaded and maintained, which by necessity is during low flow periods.

Table 4-7. Nutrient (N, P & C) concentrations of water samples collected from the four study sites over the period September 2017 to June 2018. Long term data from McCoy's Bridge are also included.

Site	Date	Total P	Total N	NPOC measured	NH_3	FRP	NOx	Chl-a
		mg/L P	mg/L N	as TOC mg/L-C	mg/L N	mg/L P	mg/L N	ug/L
Darcy's Track	23/09/2017	0.03	0.57	5.8	0.012	0.005	0.17	< 8
	19/10/2017	0.03	0.29	4.0	0.013	0.016	0.010	9
	10/11/2017	0.05	0.44	3.8	0.003	0.002	<0.001	< 8
	2/01/2018	0.03	0.37	3.3	0.004	0.003	0.00	7
	26/02/2018	0.02	0.25	2.5	0.004	0.001	0.04	< 9
	13/04/2018	0.01	0.29	2.4	0.010	0.002	0.10	< 5
	9/05/2018	0.02	0.3	3	0.007	0.002	0.11	5
	14/06/2018	0.02	0.38	3	0.016	0.003	0.17	9
Day Rd	26/09/2017	0.02	0.34	3.9	0.005	0.003	0.13	< 5
	19/10/2017	0.02	0.35	4.5	0.009	0.003	0.045	< 5
	10/11/2017	0.02	0.28	3.8	0.010	0.004	0.01	< 5
	16/01/2018	0.02	0.26	3.4	0.005	0.003	0.05	6
	27/02/2018	0.02	0.56	2.5	0.006	0.005	0.02	< 5
	13/04/2018	0.01	0.29	2.1	0.006	0.002	0.11	< 5
	9/05/2018	0.02	0.28	3.8	0.008	0.003	0.10	< 6
	14/06/2018	0.02	0.4	5.6	0.023	0.003	0.16	< 5
Loch Garry	23/09/2018	0.03	0.47	8.1	0.01	0.006	0.18	< 8
	20/10/2017	0.05	0.44	7.0	0.003	0.003	<0.001	< 11
	11/11/2017	0.05	0.42	5.7	0.004	0.003	<0.001	< 13
	2/01/2018	0.05	0.38	6.1	0.003	0.004	<0.001	13
	26/02/2018	0.03	0.26	2.4	0.002	0.002	0.00	9
	13/04/2018	0.02	0.32	3	0.004	0.002	0.08	< 5
	9/05/2018	0.02	0.32	3.3	0.003	0.002	0.10	< 5
	14/06/2018	0.03	0.41	5.2	0.010	0.004	0.15	< 6
McCoy's Bridge	24/09/2017	0.06	1.1	11	0.07	0.007	0.36	< 9
	19/10/2017	0.04	0.39	5.3	0.003	0.003	<0.001	7
	10/11/2017	0.03	0.27	5.5	0.007	0.006	0.00	< 12
	2/01/2018	0.05	0.46	6.8	0.008	0.004	0.00	< 15
	26/02/2018	0.03	0.27	2.6	0.004	0.003	<0.001	< 9
	14/04/2018	0.02	0.29	2.1	0.002	0.002	0.03	10
	9/05/2018	0.02	0.28	3.3	0.003	0.002	0.08	8
	14/06/2018	0.03	0.3	3.5	0.002	0.002	0.08	8
Long Term Mean	*Oct-04*	0.067	-	6.9	-	0.008	0.133	
Long Term Median	*to*	0.059	-	5	-	0.004	0.05	
n	*Apr-15*	493	-	456	-	493	493	

4.5.4 Statistical Modelling

As described in Section 4.2.3, a hierarchical Bayesian linear regression model, incorporating first-order auto-regression, examined the relationship of each metabolism endpoint (GPP, ER and NEP) against daily discharge (log transformed) and temperature. Predictor variables were daily discharge, water velocity and delta discharge (the difference between discharge and previous-day discharge). All four years data were used in this analysis, which again included only data that met the acceptance criteria.

The key outcomes from this modelling are summarized in the following points and the results are presented in Table 4-8, Table 4-9 and Figure 4-10:

- The best model fit was found at lag 0 (no lag), irrespective of whether the predictor was either discharge or velocity.
- Using delta discharge as the model predictor results in non-convergence of the model, suggesting, at best, a weak predictive power of delta discharge.
- Daily Total Light is stronger than Daily Average Water Temperature as a model predictor for GPP, which consistently show positive effects on all four sites.
- Water level increases (from watering actions) caused a greater drop in ER, and a greater increase in GPP, at the downstream sites (Figure 4-10).

Table 4-8. RMSE (Root Mean Square Error) of the GPP model with candidate predictors of discharge, velocity and delta discharge, for a lag of 0 to 15 days. The lowest value is the best fit.

Lag	Discharge	Velocity	Delta discharge
0	0.414	0.414	Not converging
1	10.26	0.418	
2	0.423	0.428	
3	0.434	8.361	
4	5.654	0.437	
5	0.438	5.873	
6	7.654	0.45	
7	0.448	8.433	
8	0.452	0.516	
9	0.441	1.197	
10	1.66	0.453	
11	0.447	0.446	
12	0.447	4.095	
13	0.452	6.112	
14	21.92	0.452	
15	0.451	0.446	

The lack of a best fit model for GPP with discharge at a lag greater than 0 has been found previously in this Selected Area and is most probably due to the presumed lack of significant increases in nutrient concentrations associated with these elevated flows. This conclusion is presumed due to the lack of nutrient data across a higher flow hydrograph to test this hypothesis.

The higher number of regression correlation coefficients that are statistically different from zero for light rather than temperature for GPP in Table 4-9 is possibly due to the much greater variability in daily total light whereas temperature may differ by around 10% (when expressed in degrees Kelvin). Using a Q_{10} of 2 (i.e. rate doubles for every 10 degree increase in temperature) then a GPP variation due to temperature might at most be around a factor of 4. In contrast daily light varies much more – for example, in 2017-18, Daily PAR varied from a minimum of 0.28 Es/m^2/Day up to a maximum of 10.56 Es/m^2/Day, a factor of nearly 40.

Table 4-9. Regression coefficients from Bayesian modelling of relationships between discharge and GPP or ER. Bolded values represent regressions significantly different from 0. Rho is the coefficient of the autocorrelation term.

			Discharge			Velocity		
	Predictor	Site	2.5%	median	97.5%	2.5%	median	97.5%
GPP	Light	Darcy's Track	**0.046**	**0.063**	**0.077**	**0.037**	**0.059**	**0.075**
		Day Road	**0.046**	**0.066**	**0.082**	**0.051**	**0.069**	**0.093**
		Loch Garry	**0.052**	**0.067**	**0.086**	**0.051**	**0.068**	**0.088**
		McCoy's Bridge	**0.046**	**0.061**	**0.075**	**0.046**	**0.061**	**0.076**
	Temperature	Darcy's Track	**0.047**	**0.080**	**0.105**	**0.102**	**0.132**	**0.162**
		Day Road	-0.028	0.011	0.043	-0.036	0.005	0.042
		Loch Garry	-0.003	0.021	0.046	-0.006	0.018	0.048
		McCoy's Bridge	**0.007**	**0.035**	**0.059**	**0.002**	**0.034**	**0.067**
	Rho	-	**0.894**	**0.922**	**0.952**	**0.896**	**0.931**	**0.959**
ER	Light	Darcy's Track	-0.537	-0.022	0.006	-0.042	-0.018	0.007
		Day Road	-0.288	0.033	0.069	**0.012**	**0.043**	**0.074**
		Loch Garry	-0.888	0.016	0.046	-0.003	0.022	0.047
		McCoy's Bridge	**-0.485**	**-0.047**	**-0.022**	**-0.063**	**-0.042**	**-0.021**
	Temperature	Darcy's Track	**0.039**	**0.078**	**0.103**	**0.047**	**0.081**	**0.115**
		Day Road	**0.066**	**0.116**	**0.585**	**0.069**	**0.114**	**0.161**
		Loch Garry	-1.646	0.029	0.071	-0.001	0.036	0.072
		McCoy's Bridge	-1.804	0.068	0.095	**0.045**	**0.071**	**0.095**
	Rho	-	**0.622**	**0.745**	**0.787**	**0.719**	**0.755**	**0.791**
NEP	Light	Darcy's Track	**0.013**	**0.042**	**1.047**	**0.010**	**0.035**	**0.060**
		Day Road	-1.539	0.000	0.037	-0.024	0.011	0.042
		Loch Garry	-0.007	0.025	0.987	-0.011	0.017	0.043
		McCoy's Bridge	-0.470	0.054	0.081	**0.036**	**0.059**	**0.083**
	Temperature	Darcy's Track	-0.047	-0.001	0.083	-0.042	-0.008	0.029
		Day Road	-0.180	-0.119	1.840	-0.180	-0.135	-0.086
		Loch Garry	-1.738	-0.025	0.020	-0.058	-0.024	0.013
		McCoy's Bridge	**-0.099**	**-0.037**	**-0.003**	-0.052	-0.024	0.004
	Rho	-	**0.000**	**0.761**	**0.801**	**0.733**	**0.768**	**0.803**

Figure 4-10 shows how environmental flows (including Commonwealth watering actions) affects rates of GPP and ER. This analysis explicitly examines the effect of the extra water compared to the counterfactual of no extra water. This figure shows some very consistent patterns, although all uncertainties include a ‘zero’ effect, indicating the lack of a formal statistical significance. Despite this, added environmental water slightly suppresses ecosystem respiration with the effect more prominent at the two downstream sites. In the case of GPP, environmental water appears to increase the rate – the median rates at all sites are similar and positive although there is a greater range of GPP response at the two downstream sites.

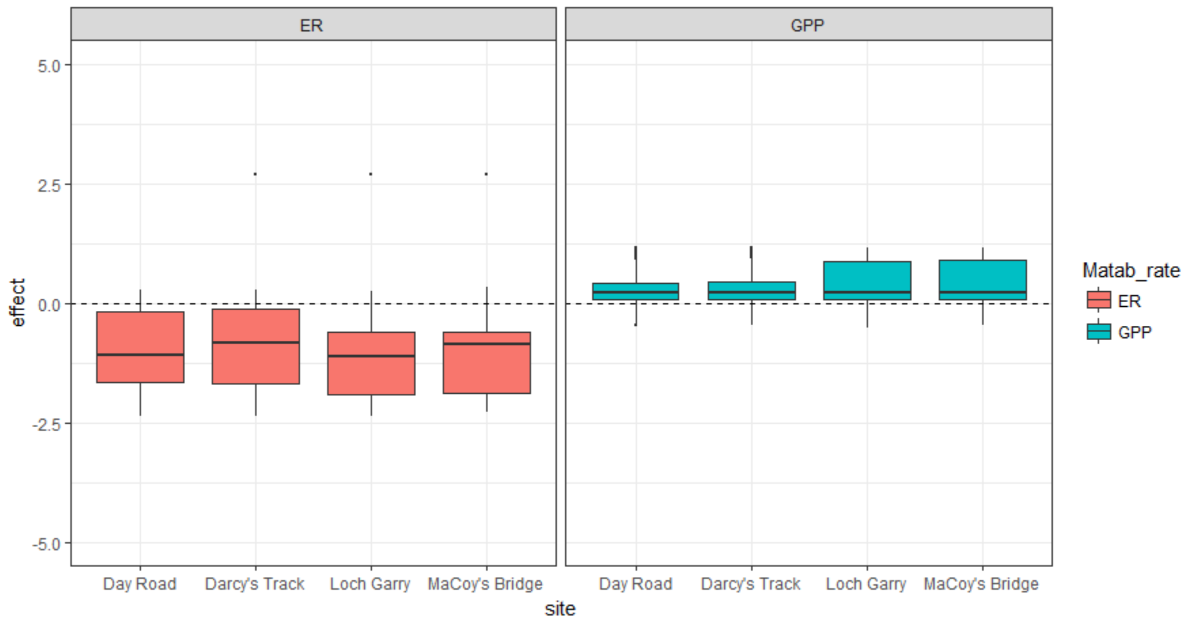


Figure 4-10. Effects of Environmental Flows (inc watering actions) on rates of ER, GPP and NEP. The error bars represent the 75% confidence intervals, summed for each site.

4.5.5 Organic Carbon Loads and Flow Categories

For the three sites (Day Rd, Darcy's Track and McCoy's Bridge) where flow categorization is possible according to Table 4-1, daily loads of organic carbon created by GPP and consumed by ER have been stratified into these categories using all four years of available data from the LTIM program. Almost all days (> 99%) with metabolic parameter estimates meeting acceptance criteria fall into three flow categories: Very Low Flow (VL), Moderately Low Flow (ML) and Low Fresh Flow (LF). The summary statistics for these daily organic carbon load data are presented in Table 4-10 (GPP) and Table 4-11 (ER). The two respective box plots are Figure 4-11 (GPP) and Figure 4-12 (ER). The Low Fresh category was omitted from these box plots and subsequent statistical testing due to the very low number of samples (n = 3).

Wilks-Shapiro tests on both raw data and common transformations (square root, log) of the raw GPP load data indicated non-normality. Consequently, Mann-Whitney Rank Sum tests were performed between each pair of flow categories at each site. All such tests showed a strong statistical difference ($p < 0.001$) for two of the McCoy's Bridge and Day Road (1 test) sites. The ML-LF comparison at McCoy's Bridge showed a moderate significant difference ($p = 0.042$).

For Darcy's Track, as shown in Figure 4-11 and Table 4-10, there was an upward trend in the median values of GPP but there were no statistically significant differences between VL and ML and between VL and LF (p = 0.69, 0.110). There was a significant difference between ML and LF ($p = 0.038$).

As with GPP, Wilks-Shapiro tests on both raw data and common transformations (square root, log) of the raw ER load data indicated failure of normality, hence, Mann-Whitney Rank Sum tests were performed between each pair of flow categories at each site. All such tests showed a strong statistical difference ($p < 0.001$) for the McCoy's Bridge (3 tests) and Day Road (1 test) sites.

For Darcy's Track, as shown in Figure 4-12 and Table 4-11, there was an upward trend in the median values but there were no statistically significant differences between VL and ML and between ML and LF (p = 0.089, 0.171). There was a significant difference between VL and LF ($p = 0.048$).

Table 4-10. Summary Statistics for Daily Organic Carbon Load (kg C/Day) created by GPP, stratified by Flow Category. All data from 2014-2018.

Site	Flow Cat	n	Mean	Median	10% Percentile	25th Percentile	75th Percentile	90th Percentile
Darcy's	Very Low	55	983	625	274	385	1310	1762
Darcy's	Mod Low	244	898	854	310	537	1220	1485
Darcy's	Low Fresh	27	1275	976	461	614	1470	1934
Day Road	Very Low	80	2546	1266	280	531	2884	7319
Day Road	Mod Low	280	4148	1950	478	1085	4677	13112
Day Road	Low Fresh	3						
McCoy's Bridge	Very Low	188	533	445	258	310	642	954
McCoy's Bridge	Mod Low	485	880	771	267	475	1110	1712
McCoy's Bridge	Low Fresh	51	1629	1670	1020	1396	1839	2277

Table 4-11. Summary Statistics for Daily Organic Carbon Load (kg C/Day) consumed by ER, stratified by Flow Category. All data from 2014-2018.

Site	Flow Cat	n	Mean	Median	10% Percentile	25th Percentile	75th Percentile	90th Percentile
Darcy's	Very Low	55	1277	954	541	606	2001	2369
Darcy's	Mod Low	244	1269	1035	349	553	1671	2508
Darcy's	Low Fresh	27	2495	1388	439	879	2890	3757
Day Road	Very Low	80	3430	1776	362	718	3610	7696
Day Road	Mod Low	280	5075	4304	745	1980	7868	10572
Day Road	Low Fresh	3						
McCoy's Bridge	Very Low	188	1298	954	529	661	1743	2464
McCoy's Bridge	Mod Low	485	1593	1399	628	1016	2056	2661
McCoy's Bridge	Low Fresh	51	1860	1539	711	1212	2515	3096

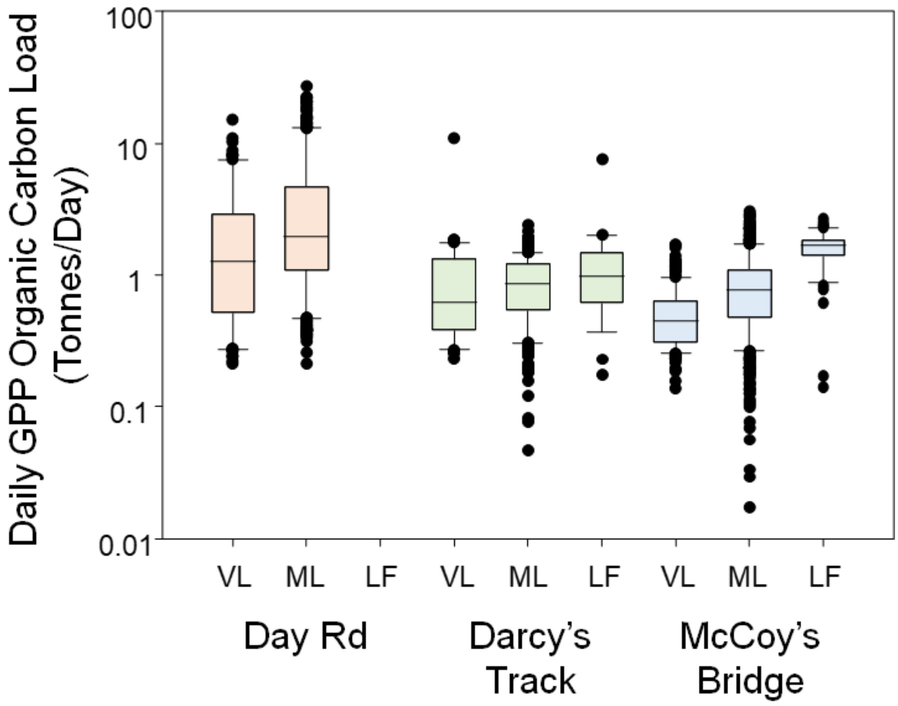


Figure 4-11. Box plot showing the Daily Organic Carbon Load (Tonnes/Day) created by GPP for all four years of LTIM data, stratified by site and flow category: VL = Very Low Flow, ML = Moderately Low Flow and LF = Low Fresh Flow. Summary statistics are presented in Table 4-10.

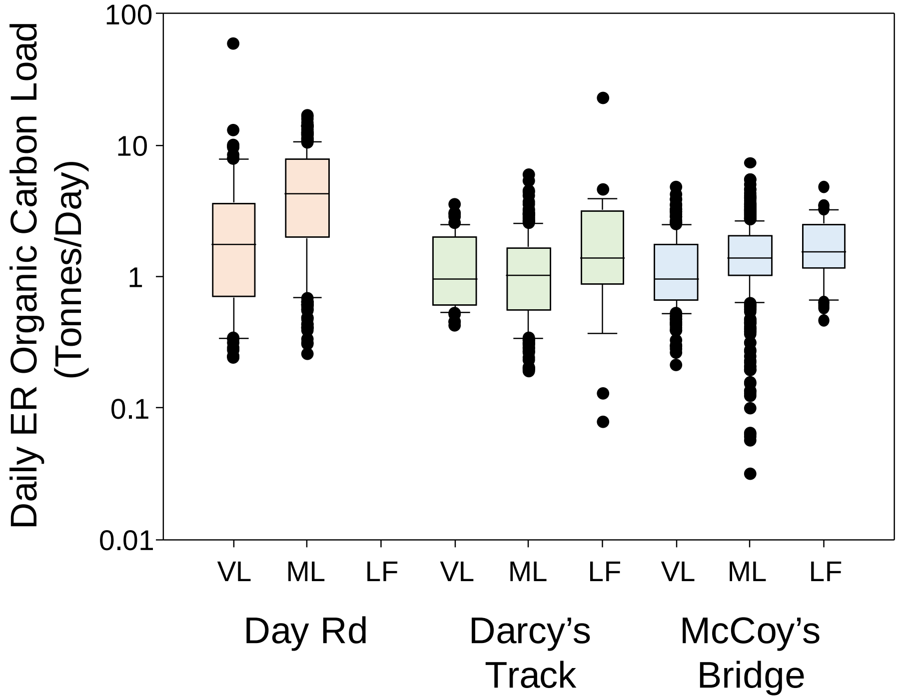


Figure 4-12. Box plot showing the Daily Organic Carbon Load (Tonnes/Day) consumed by ER for all four years of LTIM data, stratified by site and flow category: VL = Very Low Flow, ML = Moderately Low Flow and LF = Low Fresh Flow. Summary statistics are presented Table 4-11.

The key feature shown clearly in Figure 4-11 and Figure 4-12 is that increases in daily flow result in more organic carbon being created by primary production and consumed by ecosystem respiration. All three of these flow categories are for flows that are well constrained within the stream channel. This important point is developed further in the Discussion section below.

4.6 Discussion

The data presented in Figure 4-5 to Figure 4-8 do not indicate a strong relationship between GPP and flow events, consistent with findings from previous years. It is clear however that the immediate effect of flow is to lower the extant GPP (and ER) rates, almost certainly by simple dilution with large amounts of water. Primary production is expected to respond on a perhaps 10-20 day time frame following flow events (this time frame is based on typical algal doubling rates of 1-2 days), as this corresponds to sufficient time post nutrient addition to generate a significantly higher biomass of primary producers. The key assumption is that an increase in flow will introduce nutrients into the river channel which will then stimulate biomass growth and hence higher rates of GPP. It is extremely likely that the absence of significant growth is due to the extremely low bioavailable nutrient concentrations, especially the extremely low levels of filterable reactive phosphorus (which essentially equates to bioavailable phosphate). Respiration rates did seem to increase slightly in the days to weeks following discharge events. A flow-based influx of organic matter will enhance respiration although the quality/palatability of that organic matter is just as important as the increase in concentration.

Despite this expectation that there will be a lag phase between a flow event and a positive response in GPP and ER, the Bayesian modelling indicated that there was no improvement in model prediction based on discharge or velocity with any temporal lag incorporated.

The Bayesian modelling found (expected) positive relationships between GPP indicated a positive relationship between discharge (and velocity) and light at all four sites. There were significant fits at some sites between GPP and ER with temperature.

4.6.1 Impact of Daily Discharge on Stream Metabolism

Up until the end of the third year of the LTIM program, it was not clear what impact flow increases were having on rates of GPP and ER, apart from the initial decline in rates on the rising limb of the hydrograph, attributed to simple dilution by more water. This effect on GPP can again be clearly seen in Figure 4-5 with the rapid drop associated with rising discharge in early December 2017. Such examples have been highlighted in all previous annual reports. In the 2016–17 Basin Level Evaluation for Stream Metabolism, Grace (2018) introduced three derived metabolism metrics for investigating possible discharge effects, two of which are associated with daily loads of organic carbon – on a simple mass basis or mass per km of stream reach.

Using this load approach and incorporating the flow categorization of Stewardson and Guarino (2018), it was clearly demonstrated that small increases in discharge introduce more organic carbon into the stream through photosynthetic production. This is an important new finding as the initial paradigm was that no benefit to metabolism would accrue unless the water levels were sufficient to reconnect flood runners, backwaters and even the floodplain. Thus increasing flow from the very low to moderately low category means more energy (‘food’) being created to support the aquatic food web. This is a very positive finding. There is also an increase in respiration rate with flow category thus greater nutrient regeneration to sustain increased primary production.

From a management perspective, there is a positive benefit in increasing discharge, even by relatively small amounts when there are restrictions on the amount of water that can be delivered in watering actions. Nevertheless, it is likely that such increases in metabolic rates are still constrained by resources (nutrients) and much greater increases would be possible with reconnection of backwaters and floodplains.

4.7 Appendix: Logger Deployment Details

The following information relates to data absences for both the PAR loggers and DO loggers. Occasional single or two point absences were filled by linear interpolation.

4.7.1 Light

Light (PAR) was the average of the two stations (Tahbilk and Waters Road, Shepparton) except during the following periods: 04/05/2017 - 09/05/2017 (inclusive)

- Battery in light loggers ran out on 27/03/2017 at Waters Road, Shepparton and 31/03/2017 at Tahbilk. Light loggers were re-deployed on the 09/05/2017. Light data was estimated from the latitude and longitude of the sites during the period when light loggers were not collecting data. The raw PAR values calculated for each of the sites was averaged and then converted to the PAR scale of the loggers using the average conversion factor.
    - 29/08/2017 - 15/11/17 (inclusive): raw PAR at Waters Road, Shepparton stepped up from 100's - 1000's to >10,000's with maxima recorded during nighttime. Only light data from Tahbilk was used during this period.
    - On 7/03/18 PAR at Tahbilk changed -> became higher at night. Between 07/03 and 09/03/18 only PAR data from Waters Rd, Shepparton was used.

4.7.2 Darcy's Track

- Data available from 6th September 2017 until 14th June 2018.
- Downloads were performed on: 19/10/2017, 10/11/2017, 02/01/2018, 15/05/2018 and 14/06/2018.
    - Logger was not recording data between 13/10/2017 and 19/10/2017.
    - Logger was not recording data between 16/12/2017 and 02/01/2018.
    - Logger was not recording data between 06/02/2018 and 15/05/2018.
- Water level was too high to retrieve/download the logger in late Feb 2018, 13/04/2018 and 09/05/18.

4.7.3 Day Road

- Data available from 6th September 2017 until 14th June 2018.
- Downloads were performed on: 19/10/2017, 10/11/2017, 16/01/2018, 09/05/2018 and 14/06/2018.
- Water level was too high to retrieve/download the logger in late February 2018 and 13/04/2018. Data was not recorded from 31/3/18 to 9/5/18 due to battery failure.

4.7.4 Loch Garry Gauge

- Data available from 7th September 2017 until 14th June 2018.
- Downloads were performed on: 10/11/2017, 02/01/2018, 15/05/2018 and 14/06/2018.
    - No download was performed on 20/10/2017 as the logger was out of the water.
    - Logger was observed to be out of the water on 20/10/2017 and 10/11/2017. It was lowered on each occasion.
    - Logger is suspected to be out of the water during the following periods: 13-15/10/2017, 19-20/10/17 and 06-10/11/2017.
- Water level was too high to retrieve/download the logger in late February 2018, 13/04/2018 and 09/05/18. Data was not recorded from 17/3/18 to 15/5/18 due to battery failure.

4.7.5 McCoy's Bridge

- Data available from 4th May 2017 until 14th June 2018.
    - Data from June 2017, although nominally Year 3, are included here to get more wintertime data
- Downloads were performed on: 08/06/2017, 03/08/2017, 04/09/2017, 19/10/2017, 10/11/2017, 02/01/2018, 26/02/2018, 11/04/2018, 09/05/2018 and 14/06/2018.
    - Logger did not record data during deployment between 03/08/2017 and 04/09/2017.

5. Macroinvertebrates

5.1 Introduction

Macroinvertebrates are an essential part of healthy, functioning aquatic ecosystems, providing ecosystem services that range from nutrient cycling to provision of food for larger aquatic organisms such as fish. Macroinvertebrates are frequently monitored in aquatic ecosystem assessments to understand the health of those ecosystems. In the lower Goulburn River, macroinvertebrate responses have been measured to increase our understanding of how Commonwealth environmental water (CEW) affects these organisms. The aims of the macroinvertebrate monitoring program are to answer the following questions:

- What did Commonwealth environmental water contribute to macroinvertebrate diversity and abundance in the lower Goulburn River? Specifically, what combination of freshes and low flows are required to maximise macroinvertebrate abundance and biomass in the river?
- What did Commonwealth environmental water contribute to macroinvertebrate emergence in the lower Goulburn River?

In 2017–18, monitoring efforts centred on measuring macroinvertebrates before and after the second spring fresh that was delivered in mid-November, 2017. In the previous monitoring year (2016–17), the monitoring protocol had been adjusted in response to large, natural flows in spring and subsequent observations of increased crustacean productivity in response to these flows. Pre-CEW monitoring was not done due to site access issues and the fact environmental water was not delivered. Additional crustacean monitoring was conducted, which involved monthly sampling using bait traps and sweep samples from December to March, in place of the pre-CEW monitoring. The data gained from the additional crustacean surveys was highly informative, so a decision was made in 2017–18 to replace the emergence survey (yellow sticky traps) with further crustacean surveys.

5.2 Area specific evaluation questions

To determine the contribution of Commonwealth environmental water in selected areas, and to improve understanding of the relationship between specific watering actions and ecological objectives for assets, the following questions are being addressed. This information also forms the basis of Basin-scale evaluation – where area-level results are scaled up to the Basin level.

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
What did CEW contribute to macroinvertebrate diversity in the lower Goulburn River?	No	Macroinvertebrate diversity is not affected by CEW in the lower Goulburn River; in contrast, natural floods in the previous year (2016-17) did increase diversity in edge habitats. Larger natural flows would be required to have a positive impact on diversity.	Qualitative analysis of monitored results across all survey periods.
What did CEW contribute to macroinvertebrate abundance and biomass in the lower Goulburn River?	Yes	Increased flows (natural and CEW freshes) have been associated with an increase in the abundance and biomass of some taxa, particularly crustaceans such as shrimp and prawns, and possibly help sustain populations of other taxa during dry periods.	BACI ANOVAs were used to compared pre- and post-CEW data from all years (except 2016–17, which was post-flood only) to determine if abundance and biomass changed in response to the spring fresh (CEW)

5.3 Main findings from the macroinvertebrate monitoring program

5.3.1 Findings from 2017–18

The main findings from 2017–18 monitoring can be summarised as:

Artificial substrates

- Abundance did not respond to CEW, but increased at both sites post-CEW, while richness decreased in the Goulburn River post-CEW, indicating a negative effect of CEW on richness.

- Of common taxa, several showed a positive response to CEW in 2017-18, with Oligochaeta (worms) and the midge *Nanocladius* species increasing in abundance in the Goulburn River but decreasing in Broken River, while another midge (*Procladius* species) increased at both sites post-CEW, but with a much greater increase in the Goulburn.

- The midge *Tanytarsus manleyensis* showed a negative response to CEW and decreased in abundance in Goulburn River post-CEW.

- Post-CEW, biomass went from being dominated by EPT and crustaceans to being almost solely dominated by crustaceans in the Goulburn River, whereas crustacean biomass became almost negligible in the Broken River.

- Total biomass increased in the Goulburn River but decreased in the Broken post-CEW, largely due to increases in crustacean biomass.

Replicated edge sweep samples

- Abundance decreased at both sites post-CEW (not CEW related), as did number of taxa.

- Few taxa showed CEW-specific responses. Those that did were Oligochaeta (stable in the Goulburn River but decreased in the Broken River post-CEW), *Procladius* species (these became absent in RESS samples, but artificial substrate results indicate they may have shifted habitat post-CEW), *Anisops* species (became absent in the Goulburn River post-CEW) and *Paratya australiensis* (decreased in Goulburn River post-CEW).

- Total biomass was not affected by CEW (it decreased at both sites), as did biomass of major groups (crustaceans, EPT, Odonata).

Crustacean surveys: bait traps

- Freshwater prawns *Macrobrachium australiense* abundances did not immediately increase post-CEW (December) but increased in January at both Goulburn River sites. Abundances continued to increase until a slight decline in March at Loch Garry but decreased at McCoys Bridge. This may be due to an increase in complex habitats (notably vegetation) at Loch Garry that was not observed at McCoys Bridge. Dry weights largely followed abundances (so more animals meant higher dry weights), except in March, where abundances were higher than dry weights, indicating populations were made up of smaller animals.

- Changes in the abundance and dry weights of the shrimp *Paratya australiensis* did not indicate any effect of CEW on their populations in the Goulburn River. However, due to their patchy occurrence in bait traps (on average, <1 animal per trap per site each month), it is difficulty to confidently ascribe any changes in shrimp abundance or dry weights over time as significant effects.

Crustacean surveys: additional RESS

- *Paratya australiensis* abundances did not show a rapid response to CEW. It slowly increased over summer post-CEW at McCoys Bridge (while greatly increasing in the same months at Loch Garry), before dropping off in March. Post-CEW (December), biomass decreased at McCoys Bridge but this may have been due to low dissolved oxygen earlier that month rather than an effect of CEW.

- *Macrobrachium australiense* abundances were lowest at both sites in December post-CEW. This is possibly due to low dissolved oxygen because the response is not consistent with previous years. Abundances did increase in January before declining again in March. *Macrobrachium australiense* biomass was lower in December post-CEW than pre-CEW at McCoys Bridge (again possibly due to low dissolved oxygen rather than CEW), and only increased again in February before massively declining in March. Abundances were also low at Loch Garry in December but increased slightly in subsequent months.

- Immature crustaceans were most abundant at McCoys Bridge in December following the spring fresh, whereas their highest abundance was at Loch Garry in January when sustained high summer flows inundated terrestrial bank vegetation, including grasses, and potentially provided suitable habitat for this vulnerable age group of crustaceans. Abundances were very low in February and March. Their biomass followed the same patterns as abundance.

5.3.2 How these build on findings from years 1 to 3

Artificial substrates

- CEW delivered as spring freshes seems to prevent macroinvertebrate abundances and biomass of large invertebrates from decreasing as severely in the Goulburn River as they do in the Broken River post-CEW. Overall, CEW had a significant positive effect on macroinvertebrate biomass.

- Over the years of the monitoring program, several taxa have shown consistent responses to CEW spring freshes. CEW has a positive effect on Oligochaeta by reducing negative responses over time, and on *Procladius* species, with much greater positive responses in the Goulburn River post-CEW. In contrast, several species have consistent negative responses to CEW in the Goulburn River, while abundances increased, remained unchanged or were less severely reduced in the Broken River. These included *Tanytarsus manleyensis*, *Ecnomus pansus*, *Parakiefferiella* species, *Nilotanypus* species and *Rheocricotopus* species.

Replicated Edge Sweep Samples (RESS) (compared to years 2 and 3 only)

- Consistent with 2015–16, post-CEW abundance decreased at both sites; abundances were slightly higher in 2017–18 than 2016–17 (blackwater event), but actually lower post-CEW in 2015–16. Overall, CEW had a significant positive effect on macroinvertebrate abundances in RESS samples.

- Of the common taxa caught in edge habitats, most showed responses that were not consistent over the years (e.g. showed a positive response to CEW in one year but a negative response in another),or were consistent but responses appeared to be more related to site or seasonal preferences. Two taxa, Oligochaeta and *Tanytarsus manleyensis*, had overall positive responses to CEW.

- Two crustaceans have shown consistent responses to CEW delivered as spring freshes. The prawn *M. australiense* has always increased in abundance at McCoys Bridge post-CEW, but had the greatest increase in abundance post-flood (2016–17). The shrimp *P. australiensis* always has always decreased in abundance in edge habitats post-CEW; in contrast, its abundance was greatest in 2016–17 (post-flood).

- Large invertebrate biomass did not show consistent responses to spring freshes; in 2015–16, it increased at McCoys Bridge post-CEW but decreased in 2017–18. Overall, large invertebrate biomass showed significant negative responses to CEW. Biomass was greatest post-flood (2016–17) than post-CEW (2015–16 and 2017–18) due to the large increase in crustaceans present in edge habitats.

Crustacean surveys: bait traps (compared to year 3 only)

- Elevated flows in spring seem to have an important, positive effect on increasing crustacean abundances and biomass, particularly *M. australiense*. However, a comparison between the spring fresh in 2017–18 to large, natural floods in 2016–17 indicates that the magnitude of this effect is smaller with CEW compared to the overbank flood. It is worth noting, however, that there may be longer-term benefits from CEW of sustaining crustacean populations into warmer, drier months.

Crustacean surveys: additional RESS (compared to year 3 only)

- Abundances and biomasses of both *Macrobrachium australiense* and *Paratya australiensis* were not as high post-CEW in 2017–18 as they were post-flood (2016–17). They slowly increased over the summer months in 2017–18, whereas in 2016–17 they declined in January following a blackwater event. The responses are significant given the likely importance of these species in the diet of native fish.

5.4 Methods

5.4.1 Field and laboratory methods

The methods used for monitoring macroinvertebrates are given in (Webb et al. 2018), with modifications described in (Webb et al. 2017). Briefly, four methods were employed at three sites in the region: two impacted sites (Goulburn River at McCoys Bridge and Goulburn River at Loch Garry) and the control site (Broken River at Shepparton East) (See Summary Report Figure 1). The timing of monitoring, along with significant catchment events is given in Table 5-1.

The first two methods, artificial substrates and replicated edge sweep samples (RESS), were conducted at the Goulburn River at McCoys Bridge and the Broken River. Artificial substrates were adapted from (Cook et al. 2011). These are plastic mesh cylinders containing an artificial substrate (onion bags) that are deployed at each site for four to six weeks, allowing macroinvertebrates to colonise these during that time. The second method involves conducting Replicated Edge Sweep Sampling (RESS) at each site. This method is modified from that of (Gigney et al. 2007a, b) and involves taking five replicate sweep samples across the different types of edge habitat at each site. Monitoring for each method typically occurred before Commonwealth environmental water delivery (usually a spring fresh) and after environmental water.

The third and fourth methods – bait traps and additional RESS samples – specifically targeted crustaceans and were done at the two Goulburn River sites: McCoys Bridge and Loch Garry. These were conducted monthly from December to March. Twenty bait traps were deployed overnight at each site once a month. The traps were placed among four habitat types (bare, coarse organic particulate matter/depositional areas, macrophytes and snags). Upon retrieval, all crustaceans were removed from the bait traps and stored in 100% ethanol with the exception of yabbies (*Cherax* species), which were counted, weighed and released back into the river. The preserved crustaceans were identified to species in the laboratory and had their carapace lengths measured (from the tip of the rostrum to the end of the carapace). These were air dried for 24 hours, dried in the oven at 60°C for a further 24 hours and weighed. Additional RESS samples were taken from both Loch Garry and McCoys Bridge when bait traps were being retrieved using a modified version of the original RESS method. Modifications included measuring the area swept during the survey so that biomass could be calculated and preserving the whole sample to ensure small crustaceans (larvae) would not be missed. Samples were preserved in 100% ethanol and crustaceans were picked from these in the laboratory. Crustaceans from RESS samples were also identified, measured, dried and weighed for biomass.

Table 5-1. Macroinvertebrate sampling times and significant events in the Goulburn and Broken Rivers during 2017-18. CEW = Commonwealth Environmental Water delivered as spring freshes. Pre-CEW = pre-Commonwealth Environmental Water delivery (before spring fresh); Post-CEW = post-Commonwealth Environmental Water delivery (after spring fresh); GM =

Goulburn River at McCoys Bridge; GL = Goulburn River at Loch Garry; BR = Broken River at Shepparton East. D = deployed; R = retrieved.

Activity / event	Site	Sampling dates Sep-17	Oct-17	Nov-17	Dec-17	Jan-18	Feb-18	Mar-18
Events	Goulburn River	CEW1 start 16/9	CEW1 end 7/10	CEW2 start 19/11	CEW 2 end 1/12; high flows and low dissolved oxygen 3/12-6/12	Elevated flows for consumptive demand	Elevated flows for consumptive demand	Elevated flows for consumptive demand
RESS	GM		Pre-CEW 11/10			Post-CEW 9/1		
	BR		Pre-CEW 12/10			Post-CEW 10/1		
Artificial substrates	GM		Pre-CEW 11/10 D	Pre-CEW 6/11 R		Post-CEW 9/1 D	Post-CEW 6/2 R	
	BR		Pre-CEW 12/10 D	Pre-CEW 7/11 R		Post-CEW 10/1 D	Post-CEW 7/2 R	
Bait traps	GM				19/12-20/12	23/1-24/1	27/2-28/2	27/3-28/3
	GL				19/12-20/12	23/1-24/1	27/2-28/2	27/3-28/3
RESS (crustaceans)	GM				20/12	24/1	28/2	28/3
	GL				20/12	24/1	28/2	28/3

5.4.2 Statistical analysis

BACI ANOVAs within a Bayesian framework were also used to assess the effect of the spring fresh on macroinvertebrate biomass and abundance (as measured using artificial substrates and replicated edge sampling). The model is structured as follows:

$$y_i \sim Normal(mu_{i,} s^2)$$

$$mu_i = eff.Year_m + g.mu_{b,r}$$

The macroinvertebrate biomass/abundance (*y*) (fourth-root transformed) in sample *i* is normally distributed, with a mean of mu_i and standard deviation of *s*. mu_i is driven by the global abundance at river *r* (Broken River or Goulburn River) and at time *b* (before or after the spring fresh) (*g.mu*), and the random effect of year (*eff.Year)*.

The global biomass/abundance (*g.mu*) is drawn from a truncated normal distribution with a mean of 0, standard deviation of *s.g.mu* and a minimum of 0. Likewise, the random effect of year (*eff.Year)* is drawn from a normal distribution with a mean of 0 and standard deviation of *s.year.*

$$g.mu_{r,b} \sim Normal(0, s.g.mu^2)I(0,)$$

$$eff.year_m \sim N(0, s.year^2)$$

5.5 Results

5.5.1 Artificial substrates

In 2017–18 (Year 4), a total of 17,850 macroinvertebrates from 107 taxa were caught in artificial substrates. Taxonomic richness was lower in the Goulburn River than the Broken River and decreased post-CEW in the Goulburn but not the Broken (Figure 5-1a). These post-CEW changes in the number of taxa were not consistent over the years and do not appear to be related to CEW spring fresh delivery. Total macroinvertebrate abundance was consistently higher in the Broken River than in the Goulburn River, but in both rivers it increased post-CEW (Figure 5-1b). When post-CEW changes in abundance were compared across years (when CEW was delivered as spring freshes), macroinvertebrate abundances decreased at both sites, but the decrease was less severe in the Goulburn River (Figure 5-1c). However, the effect of CEW on abundance was not significant (Table 5-2). Abundances post-CEW in all years tended to be higher than post-flood (2016–17), where abundances in artificial substrates were reduced due to a blackwater event.

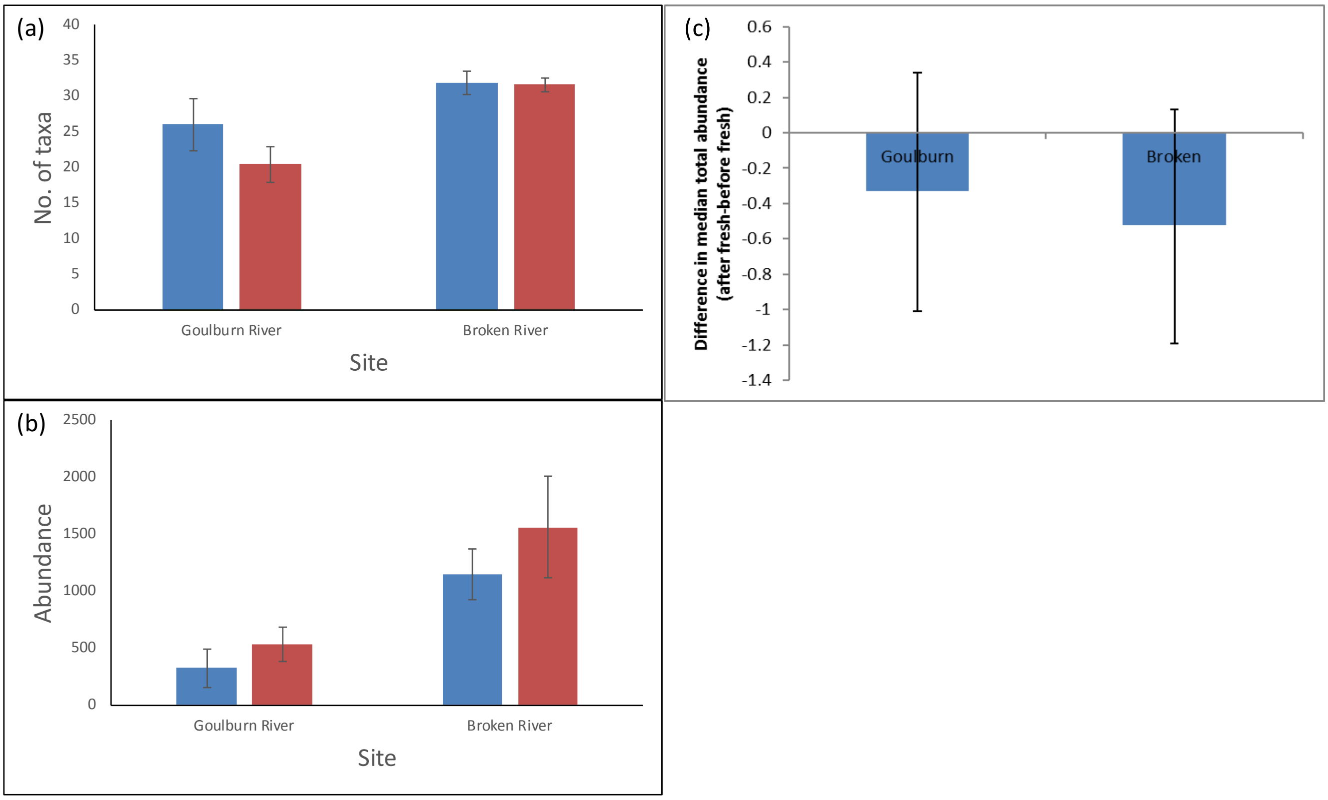


Figure 5-1. (a) Number of taxa (average ± standard error of the mean) and (b) abundance (average ± standard error of the mean) of macroinvertebrates in artificial substrates from 2017-18 pre-CEW (blue) and post-CEW (red), and (c) change in total median abundance (post-CEW minus pre-CEW) of macroinvertebrates across all years except 2016-17 (± 95% Bayesian credible intervals).

There were 15 common taxa that each contributed to >1% of the total abundance; these were compared to common taxa from previous years to determine a final list of taxa for further analyses. Ten taxa were consistently common across all four sampling years and were considered further (Table 5-2). Ignoring data from 2016–17 (post-flood only), several taxa showed relatively consistent responses across the years. For example, *Nilotanypus* species generally increased in abundance at both sites post-CEW, although this response indicates abundance increased with warming temperatures as summer approached as opposed to an effect of CEW (Table 5-2). *Nilotanypus* species also showed a strong preference for site and was consistently more abundant in the Broken River than the Goulburn River. *Procladius* species also increased in abundance post-CEW at both sites, but the increase was much greater in the Goulburn River suggesting the spring fresh had a positive effect on this species. In general, most species were more abundant post-CEW in all years than post-flood (2016–17) in the Goulburn River, after that rive experienced low dissolved oxygen during a blackwater event while the substrates were deployed.

Table 5-2. Average abundance of common taxa pre- and post-Commonwealth Environmental Water (CEW) delivery as spring freshes in 2014-15, 2015-16 and 2017-18, along with post-flood abundances in 2016-17 at two sites in the lower Goulburn River. GR = Goulburn River. BR = Broken River.

		Average abundance						
Taxon	Site	Pre-CEW 2014-15	Post-CEW 2014-15	Pre-CEW 2015-16	Post-CEW 2015-16	Post-flood 2016-17	Pre-CEW 2017-18	Post-CEW 2017-18
Rheotanytarsus species	GR	1	11	0	1	1	2	0
	BR	9	54	37	<1	133	62	511
Oligochaeta	GR	273	46	3	3	70	85	123
	BR	456	54	4	3	172	187	57
Nilotanypus species	GR	2	7	1	1	0	0	4
	BR	24	55	1	3	90	10	202
Nanocladius species	GR	177	87	56	74	6	17	33
	BR	2	23	15	3	5	148	20
Procladius species	GR	6	175	12	167	17	29	140
	BR	1	7	1	14	11	5	17
Tanytarsus manleyensis	GR	39	32	95	70	0	40	7
	BR	12	23	24	12	0	13	51
Ecnomus pansus	GR	11	37	2	3	3	1	4
	BR	10	75	2	3	77	5	58
Parakiefferiella species	GR	12	<1	262	27	<1	8	0
	BR	25	3	48	15	5	21	<1
Ceratopogonidae	GR	9	9	2	2	17	6	8
	BR	6	5	1	1	12	11	21
Rheocricotopus species	GR	49	2	0	1	<1	5	1
	BR	7	2	0	0	3	0	40

The differences in abundance between pre- and post-CEW for these common taxa over the years when CEW was delivered were analysed using BACI ANOVAs and the results are given in Table 5-3. *Rheotanytarsus* species abundances decreased in the Goulburn River post-CEW while it increased in the Broken River but responses to CEW were not significant (Figure 5-2a). Oligochaeta abundance decreased at both sites post-CEW, but with much greater decreases in the Broken River, indicating CEW had a significant beneficial effect on these organisms (Figure 5-2b). *Nilotanypus* species abundances increased at both sites post-CEW, but the increase in abundance was greater in the Broken River (Figure 5-2c). As a result, CEW was found to have a significant negative effect on *Nilotanypus* species compared to when CEWs were absent, but the difference in abundance between sites probably relates to the preference of this genus for sandy streams rather than an effect of CE . *Nanocladius* species abundances decreased at both sites post-CEW, with a similar difference between sites showing no significant effect of CEW (Figure 5-2d). *Procladius* species abundances increased post-CEW at both sites, with a greater increase in the Goulburn River that indicated a statistically significant positive effect of CEW (Figure 5-2e). *Tanytarsus manleyensis* abundances decreased in the Goulburn River but increased in the Broken River post-CEW, indicating the spring fresh had a significant negative effect on this species (Figure 5-2f). Similarly, *Ecnomus pansus* abundances decreased in the Goulburn River but increased in the Broken River post-CEW, again indicating a significant negative effect of the spring fresh (Figure 5-2g). *Parakiefferiella* species abundances decreased at both sites post-CEW, but with much greater decreases in the Goulburn River (Figure 5-2h). As a result, CEW was associated with a significant negative effect on this genus. Ceratopogonidae abundances increased at both sites post-CEW, but the increase in abundance was greater in the Broken River and there was weakly significant evidence of a negative effect of CEW on this family (Figure 5-2i). *Rheocricotopus* species abundances decreased in the Goulburn River but increased in the Broken River post-CEW, indicating a significant negative effect of spring fresh (Figure 5-2j).

Table 5-3. Posterior probability of effects (both positive and negative) of CEW obtained by the differences in the before-after effect in the Goulburn and Broken Rivers in artificial substrates. Values closer to 1 – significant positive effect; values closer to 0 – significant negative effect; values closer to 0.5 – insignificant differences. We set the significance threshold at 0.75, 0.25. Significant positive effects are shaded green while significant negative effects are shaded red.

	Mean
Total abundance	0.65
Total biomass	0.78
Oligochaeta	0.90
***Rheotanytarsus* species**	0.57
Tanytarsus manleyensis	0.13
***Procladius* species**	0.91
Ecnomus pansus	0.12
***Parakiefferiella* species**	0.04
***Nanocladius* species**	0.51
***Nilotanypus* species**	0.02
Ceratopogonidae	0.22
***Rheocricotopus* species**	0.02

In 2017-18, total large invertebrate biomass (invertebrates >5mm) increased in the Goulburn River but decreased in the Broken River post-CEW (Figure 5-3a). The change in biomass, when considered with the other years when CEW was delivered as spring freshes, was usually seen as a decrease in biomass post-CEW at both sites, with a much greater decrease in the Broken River indicating CEW may have had some positive impact on biomass (Figure 5-3b). As a result, CEW delivered as a spring fresh has a significant positive effect on large invertebrate biomass (Table 5-3).

A breakdown of the main taxonomic groups contributing to biomass in 2017–18 showed distinct site and event differences. In the Goulburn River, the contribution of crustaceans to biomass increased post-CEW, while in the Broken River their contribution decreased while Ephemeroptera, Plecoptera and Trichoptera (EPT) dominated biomass (Figure 5-3c), not because they increased in biomass (Figure 5-3d) but because crustacean biomass was heavily reduced post-CEW at this site (Figure 5-3e). Odonata biomass decreased in the Goulburn River but increased in the Broken River post CEW (Figure 5-3f), while other taxa contributed relatively little to large invertebrate biomass.

5.5.2 Replicated Edge Sweep Samples (RESS)

A total of 1,962 individuals from 87 taxa were identified in RESS samples, with abundance and taxonomic richness always higher in the Broken River than in the Goulburn River. Both abundance and richness decreased post-CEW at both sites (Figure 5-4a,b), which is similar to 2015–16. The spring freshes had a significant positive effect on total abundance in RESS samples (Table 5-4), due to the decrease in abundance at both sites post-CEW being much less in the Goulburn River (Figure 5-4c).

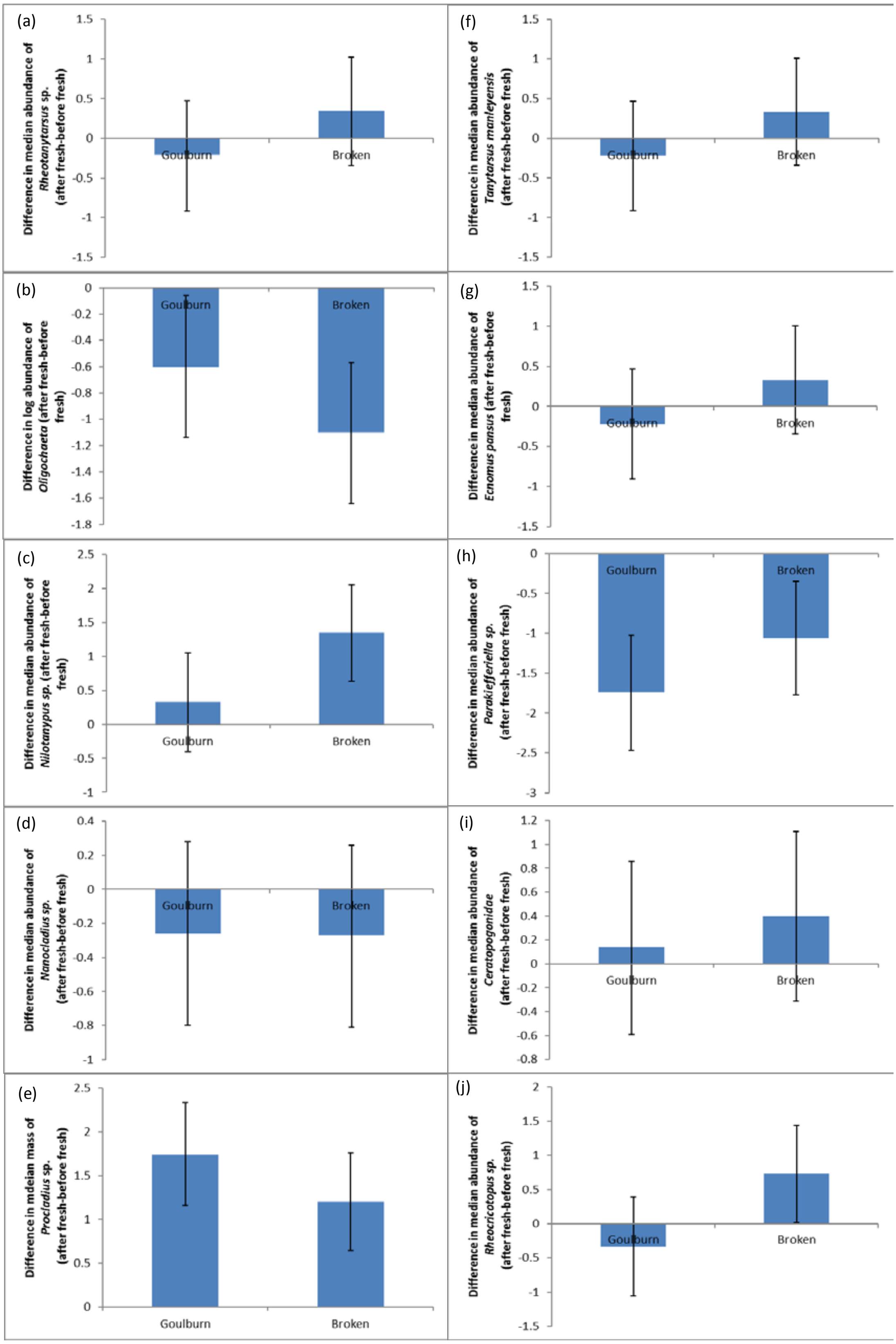


Figure 5-2. Change in median abundance of (a) *Rheotanytarsus* species, (b) Oligochaeta, (c) *Nilotanypus* species, (d) *Nanocladius* species, (e) *Procladius* species, (f) *Tanytarsus manleyensis*, (g) *Ecnomus pansus*, (h) *Parakiefferiella* species, (i) Ceratopogonidae and (j) *Rheocricotopus* species (post-CEW minus pre-CEW) in artificial substrates. Error bars indicate the 95% Bayesian credible intervals.

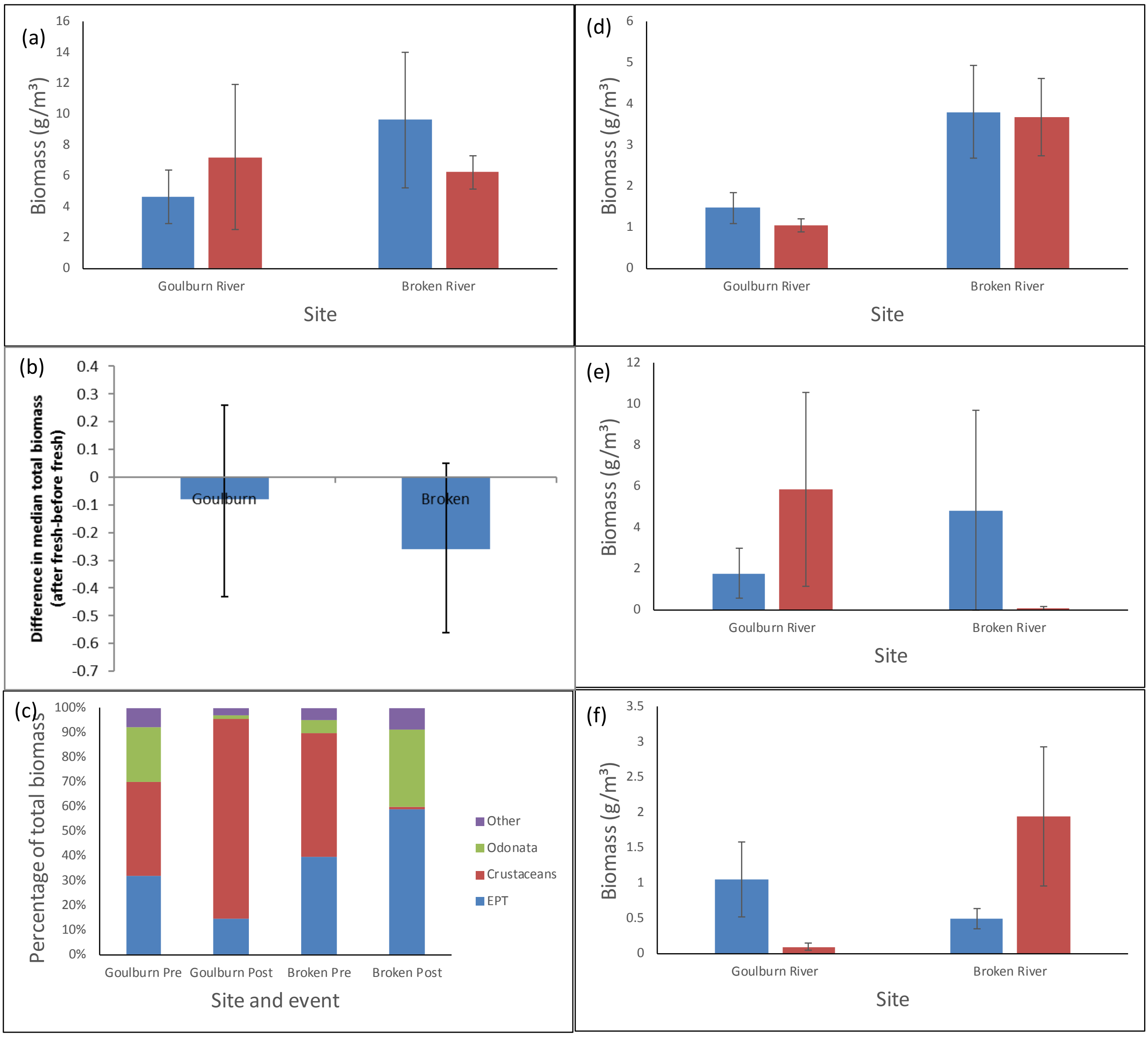


Figure 5-3. Biomass in artificial substrates. (a) Average total large invertebrate biomass in 2017–18 (+ standard error of the mean). (b) Change in median total biomass across all years except 2016–17 (post-CEW minus pre-CEW; error bars are 95% Bayesian credible intervals). (c) Percentage contribution of main large invertebrate groups to total biomass in 2017-18. Average (+ standard error of the mean) biomass in 2017-18 of (d) Ephemeroptera, Plecoptera and Trichoptera (EPT), (e) crustaceans and (f) Odonata. For figures (a), (d), (e) and (f) blue = pre-CEW, red = post-CEW.

Responses of specific taxa to CEW in RESS samples were examined further. These taxa were selected based on those that were also most abundant in artificial substrates. Significant effects of CEW delivery were examined by comparing changes in abundance (after CEW minus before CEW) from both sites over the years (Table 5-4). Four taxa showed significant positive responses to CEW. Although Oligochaeta responses to CEW were not consistent over the years (Table 5-5), in general they showed a positive response to CEW compared to when they were not exposed to CEW in the Broken River. The Chironomidae *Rheotanytarsus* species and *Tanytarsus manleyensis* also had significant positive responses to CEW. However, in the case of *T. manleyensis* this was attributed to a less severe decrease in abundance in the Goulburn River post-CEW compared to the Broken River, a decrease that is possibly seasonal (Table 5-5). Meanwhile, Ceratopogonidae abundances increased post-CEW in the Goulburn River but decreased in the Broken River, but the posterior probability of the change was not significant.

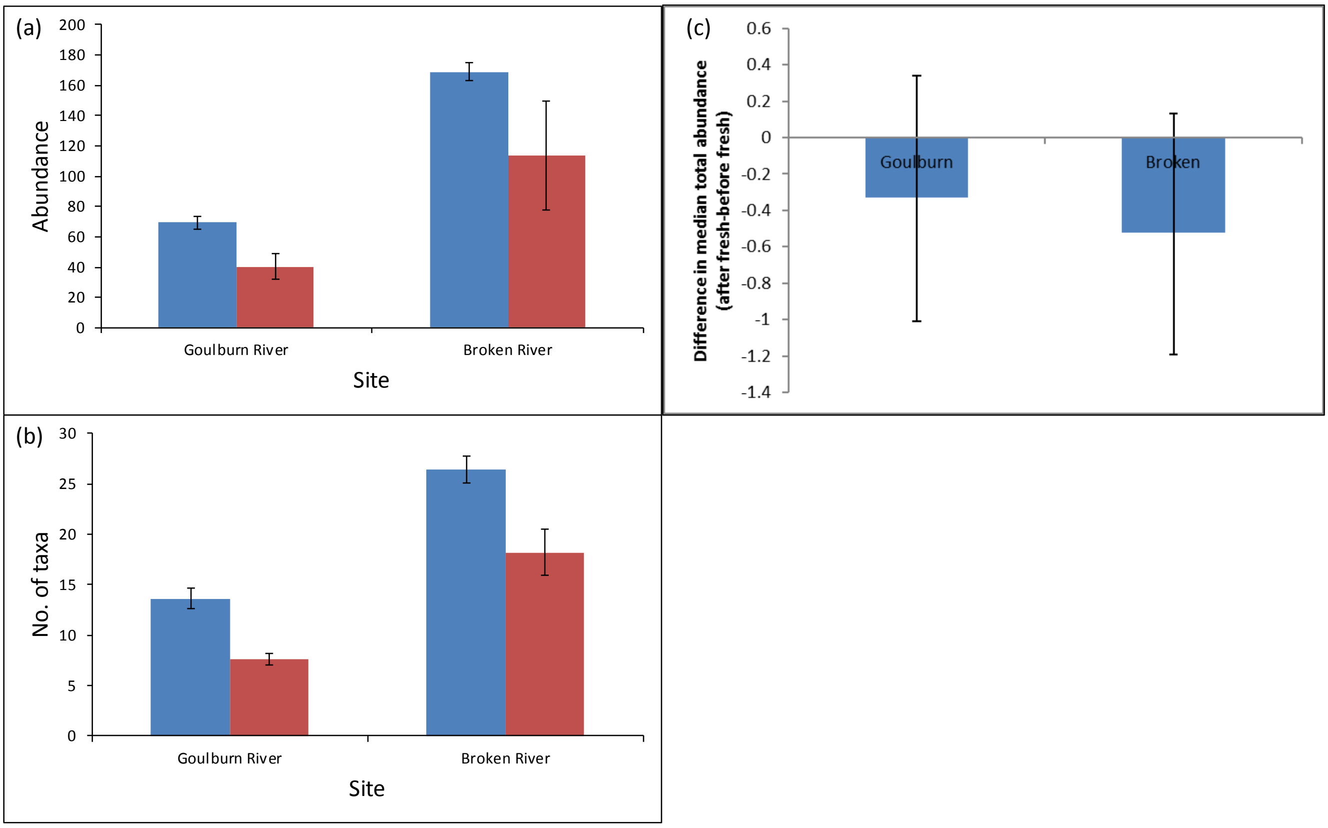


Figure 5-4. Average (+ standard error of the mean) (a) abundance, (b) number of taxa in replicated edge sweep samples from 2017-18 and (c) change in median total abundance (post-CEW minus pre-CEW). For figures (a) and (b), blue columns = pre-CEW and red columns = post-CEW. In figure c, data are 4th-root transformed and error bars indicate the 95% Bayesian credible intervals.

Several taxa showed overall significant negative responses to CEW in the Goulburn River (Table 5-4). Unlike in artificial substrates (where a positive response was observed), in RESS samples *Procladius* abundances consistently decreased post-CEW in the Goulburn River (although consistent responses were not observed in the Broken River (Table 5-5). Similarly, the caddisfly *Ecnomus pansus* showed negative responses to CEW, as did the Chironomidae *Parakiefferiella* species, *Nilotanypus* species and *Rheocricotopus* species. *Nanocladius* species did not significantly respond to CEW.

There were 19 common taxa that contributed to >1% of the abundance in 2017–18; these were compared to common taxa in previous years to derive a list of common taxa that were considered further. Common taxa (considered across all years) are listed in Table 5-5, along with changes in their abundance between post-CEW and pre-CEW at both sites for the two years when CEW was delivered as a spring fresh (2015–16 and 2017–18). Post-flood data (from 2016–17) was not considered in this section. Results were compared across the two years to determine consistent responses that are indicative of an effect of the spring freshes (CEW). For responses to be attributed to CEW, it had to occur in both years (2015–16 and 2017–18) and had to result in a change in abundance in the Goulburn River that did not occur in the Broken River. Other consistent effects were also observed, notably seasonal changes (e.g. a taxon always increased or decreased in abundance post-CEW regardless of site) or site preferences (i.e. a taxon was consistently present at one site and absent from the other).

Table 5-4. Posterior probability of significant effects of CEW obtained by the differences in the before-after effect in the Goulburn and Broken Rivers in RESS samples. 1 – significant positive effect; 0 – significant negative effect; 0.5 – insignificant differences. Significant positive effects are shaded green while significant negative effects are shaded red.

	Mean
Total abundance	0.98
Total biomass	0.08
Oligochaeta	1.00
***Rheotanytarsus* species**	0.84
Tanytarsus manleyensis	0.99
***Procladius* species**	0.00
Ecnomus pansus	0.10
***Parakiefferiella* species**	0.16
***Nanocladius* species**	0.32
***Nilotanypus* species**	0.07
Ceratopogonidae	0.71
***Rheocricotopus* species**	0.12

While several common taxa did show consistent effects across years, these were often due to site preferences (i.e., *Offadens confluens* and *Caridina indistincta* were only found in the Broken River) or seasonal preferences (e.g. *O. confluens* and *Tanytarsus manleyensis* were consistently less abundant during warmer, post-CEW sampling regardless of whether a site experienced CEW or not) (Table 5-5). Only two taxa showed consistent responses to CEW delivery. *Macrobrachium australiense* increased in abundance in the Goulburn River post-CEW, while these responses were not consistent in the Broken River. In contrast, *Paratya australiensis* decreased in abundance post-CEW in the Goulburn River. Further evidence that these two crustaceans were responding to increased flows came by comparing post-CEW data from 2015–16 and 2017–18 to post-flood data from 2016–17. In 2016–17, both *M. australiense* and *P. australiensis* abundances were much higher in RESS samples post-flood than post-CEW in other years.

When compared across years, CEW was shown to have significant negative effects on large invertebrate biomass in RESS samples (Table 5-4), with biomass generally decreasing post-CEW in the Goulburn River but increased in the Broken River (Table 5-5). In 2017-18, large macroinvertebrate biomass decreased at both sites post-CEW (Figure 5-6a).

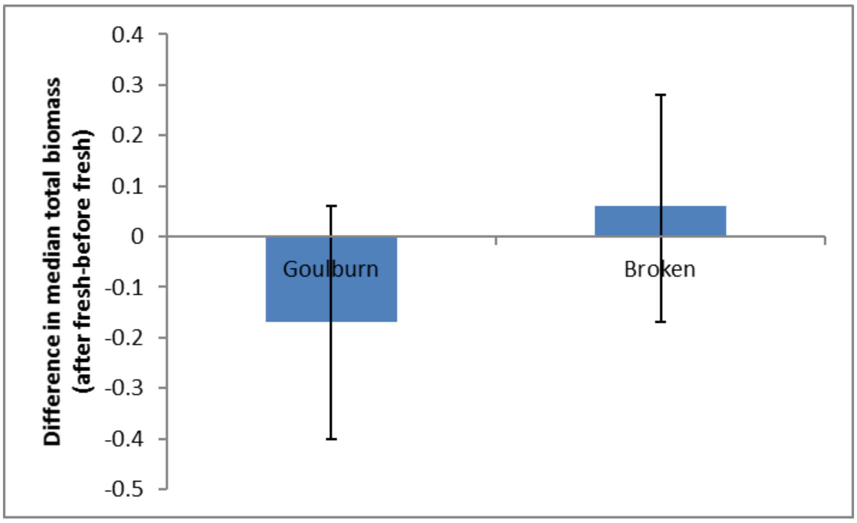


Figure 5-5. Change in median biomass (post-CEW minus pre-CEW) in replicated edge sweep samples. Data were 4th-root transformed. Error bars indicate the 95 percent Bayesian credible intervals.

Table 5-5. Common taxa from replicated edge sweep samples, changes in their abundance (post-CEW – pre-CEW) and what consistent changes might mean.

Taxon	Changes in abundance (post-CEW – pre-CEW) Goulburn River 2015-16	Broken River 2015-16	Goulburn River 2017-18	Broken River 2017-18	Consistent effects?
Micronecta annae	↓	↓	↑	↑	No
Offadens confluens	Absent	↓	Absent	↓	Yes; preference for Broken and seasonal ↓
Macrobrachium australiense	↑	↓	↑	↑	Yes; ↑ after CEW
Caridina indistincta	Absent	Unchanged	Absent	↑	Yes; preference for Broken
Oligochaeta	↓	↓	↑	↓	No
Procladius species	↓	↓	↓	↑	No
Tasmanocoenis tillyardi	↓	↑	↓	↓	No
Anisops species	↑	Absent	↓	Unchanged	No
Tanytarsus manleyensis	↓	↓	Absent	↓	Yes; seasonal ↓
Paratya australiensis	↓	Absent	↓	Unchanged	Yes; ↓ after CEW
Triaenodes species	Absent	↓	Unchanged	↓	No
Cryptochironomus species	↓	↓	Absent	Absent	No
Atalophlebia species AV6	Unchanged	↑	↓	↓	No

Regardless of site or sampling event, in 2017–18 large macroinvertebrate biomass was dominated by crustaceans, with other major groups contributing little to biomass (Figure 5-6b). However, the biomass of all three major large invertebrate groups, along with other taxa, decreased post-CEW at both sites (Figure 5-6c-f).

5.5.3 Additional crustacean surveys: bait traps

Abundance and dry weights

As in the previous year, *Macrobrachium australiense* were the most abundant crustaceans caught in bait traps. At both McCoys Bridge and Loch Garry their abundance increased from December to January (Figure 5-7a). It continued to increase over summer at Loch Garry, while it decreased in February and March at McCoys Bridge. A similar pattern was observed with *M. australiense* dry weights (Figure 5-7b). The 2017-18 results were very different from the 2016–17 results, where abundances and dry weights were much greater in December after large natural floods in spring, followed by steep declines in January after a blackwater event around New Year (Figure 5-8; middle top and bottom charts). *Macrobrachium australiense* showed no preference for habitat types at either site, similar to 2016–17 (Figure 5-7c-d).

Paratya australiensis were much less common in bait traps. Although it appeared that abundances dropped off in January and March at both Goulburn River sites (Figure 5-9a). Again, changes in *P. australiensis* dry weights closely followed changes in abundance (Figure 5-9b). As in 2016–17, their patchy occurrence in bait traps made it difficult to confidently discern any temporal patterns and observe any responses to flows, both natural and CEW (Figure 5-8; top right and bottom right charts). *Paratya australiensis* showed no consistent preference for habitat types at either site (Figure 5-9c-d), which differs from 2016–17, when they were more abundant in bait traps placed in macrophytes at McCoys Bridge.

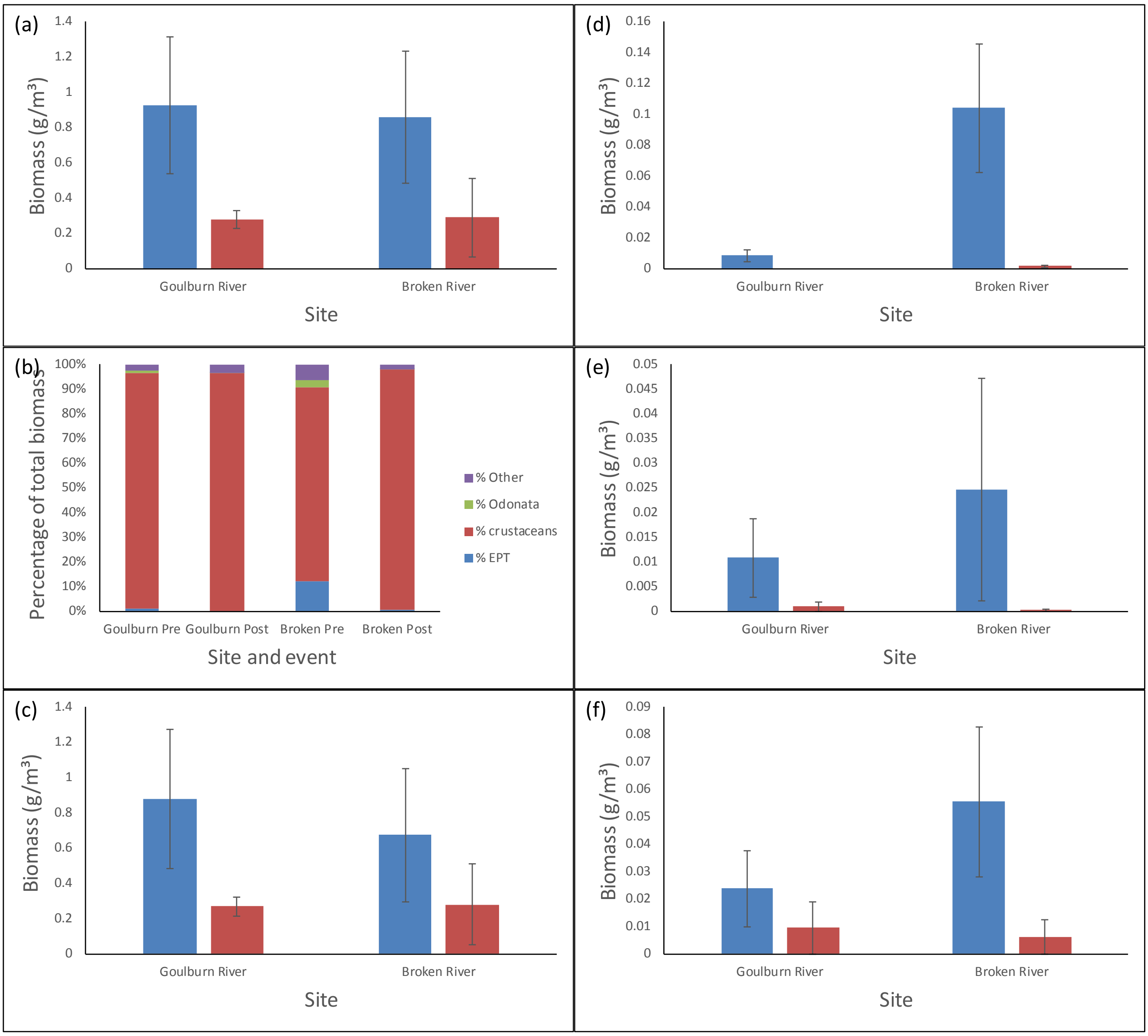


Figure 5-6. (a) Total large invertebrate biomass, (b) percentage of total biomass by major groups, (c) crustacean biomass, (d) EPT biomass, (e) Odonata biomass and (f) other large invertebrate biomass in RESS samples from 2017-18. For figures (a) and (d) to (f), values are average ± standard error of the mean, with blue columns = pre-CEW and red columns = post-CEW.

Cherax species were not very abundant in the bait traps at either site (on average, <1 animal per trap), but being larger animals when they did occur they contributed substantially to crustacean weights. They were most abundant with greatest wet weights in bait traps during December, then decreased in abundance as the months progressed at both sites (Figure 5-10). Responses at both Loch Garry and McCoys Bridge were similar in both 2016-17 and 2017-18, demonstrating that this species may have been less impacted by blackwater events due to behavioural adaptations (although it was never very abundant in bait traps, making it difficult to determine if this was actually the case) (Figure 5-8; top left and bottom left charts).

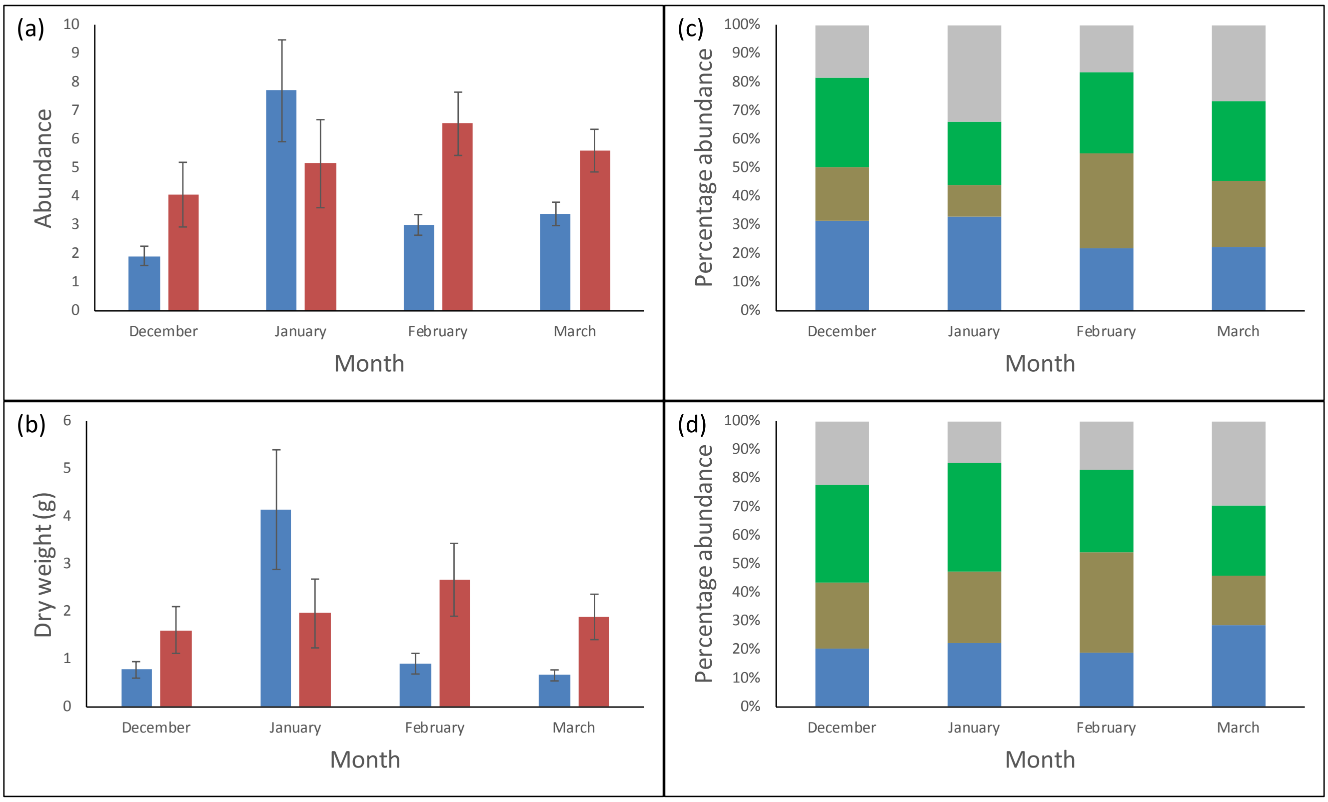


Figure 5-7. *Macrobrachium australiense* in bait traps from 2017–18. (a) average (+ standard error of the mean) abundance, (b) average (+ standard error of the mean) dry weights, (c) percentage of abundance across traps placed in different habitats at McCoys Bridge and (d) percentage of abundance across traps placed in different habitats at Loch Garry. In figures (a) and (b), blue columns = McCoys Bridge and red columns = Loch Garry. In figures (c) and (d), blue = bare habitat, brown = coarse organic particulate matter/depositional area, green = macrophytes and grey = snags.

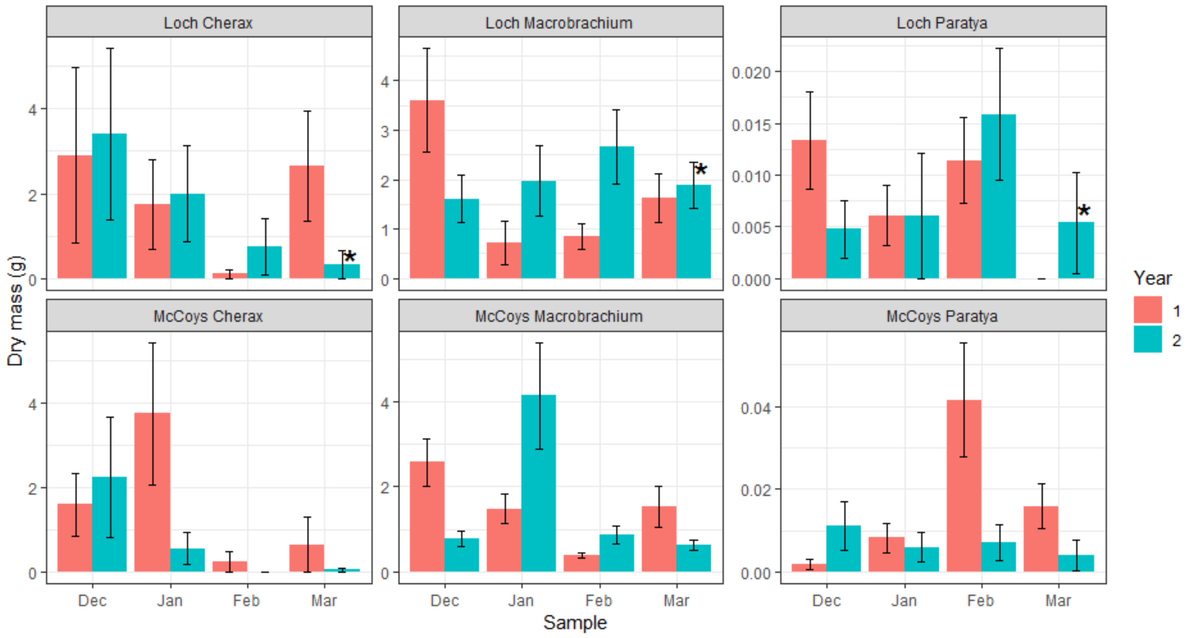


Figure 5-8. Average dry weights of crustaceans across monthly sampling over two years (Year 1: 2016-17; Year 2: 2017-18) at each site in bait traps. The top three figures are from Loch Garry while the bottom three are McCoys Bridge. Left figures = *Cherax* species; centre figures = *Macrobrachium australiense*; right figures = *Paratya australiense*. Whiskers indicate sampling errors. N = 20 at each site (except Loch Garry in March, Year 2, where N = 18).

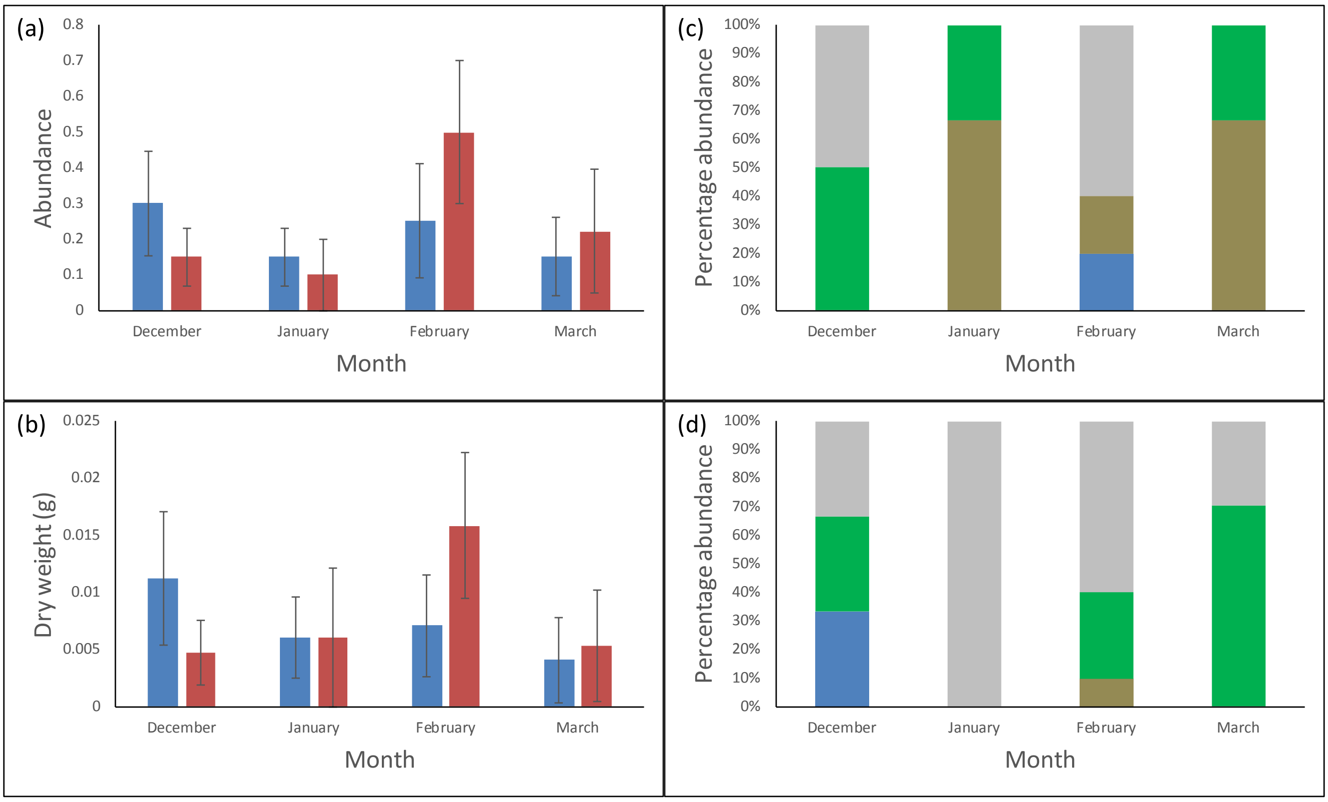


Figure 5-9. *Paratya australiensis* in bait traps from 2017-18. (a) average (+ standard error of the mean) abundance, (b) average (+ standard error of the mean) dry weights, (c) percentage of abundance across traps placed in different habitats at McCoys Bridge and (d) percentage of abundance across traps placed in different habitats at Loch Garry. In figures (a) and (b), blue columns = McCoys Bridge and red columns = Loch Garry. In figures (c) and (d), blue = bare habitat, brown = coarse organic particulate matter/depositional area, green = macrophytes and grey = snags.

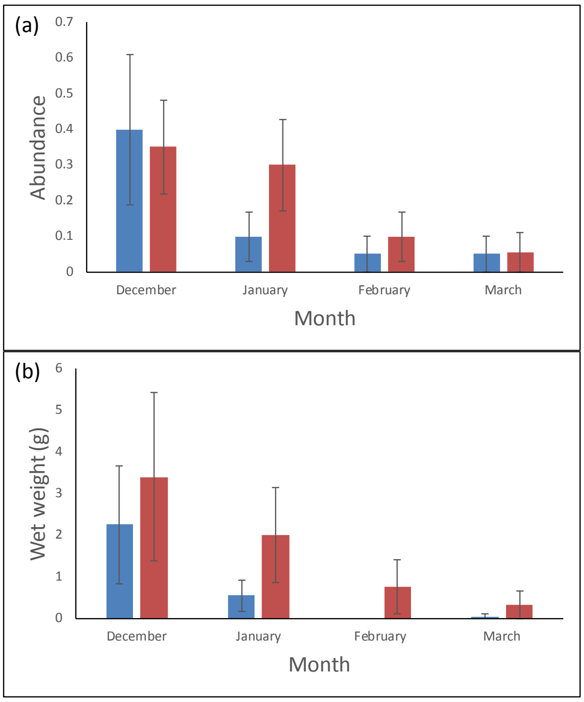


Figure 5-10. *Cherax* species (a) abundance and (b) wet weights in 2017-18. Values are averages (+ standard error of the mean). Blue columns = McCoys Bridge and red columns = Loch Garry.

Carapace lengths

Although the average carapace lengths of *M. australiense* caught in bait traps differed across sites and months, the variability in size (maximum and minimum carapace lengths, indicated as error bars on the chart) were relatively consistent between sites across months. Animals caught in December at both sites tended to be slightly larger than those in later months, as shown by both the average carapace length and the larger minimum carapace lengths (Figure 5-11a). In later months, animals with smaller carapaces (lower minimum carapace lengths) were caught in the bait traps, and average carapace lengths were somewhat smaller in February and March, indicating a shift in the size of animals making up the population. It is interesting to note that across both sites and all months, the maximum carapace length remained fairly constant, probably reflecting the maximum size of individuals in the population. Average carapace lengths tended to be lower across both sites and in all months in 2016–17 than 2017–18, except in March.

The carapace lengths of individual *M. australiense* were also assessed based on habitat types where bait traps were placed. In 2017–18, carapace lengths (averaged across site, month and habitat type) were nearly always greatest in macrophytes, while they were usually smallest in either CPOM or snags. There were also some differences between months for smallest average carapace lengths. At both sites, the smallest averaged carapace lengths tended to be in bare habitats and CPOM/depositional areas, whereas in February they were in snags. These results differ from 2016–17, where the largest and smallest averaged carapace lengths did not appear to be associated with any habitat type. In addition, there was little consistency between sites for any given month except in February, where the largest average carapace lengths were from bare habitats, while in March they were in macrophytes.

The habitats where the smallest and largest individuals (based on carapace length) were caught in each month at each site were also noted. In 2017–18, the largest *M. australiense* at each site and in each month was caught in bait traps placed near macrophytes on all occasions except one. In contrast, the habitat type where the largest individuals were caught in 2016–17 was much more variable between sites and months, although they were somewhat more likely to be present in bare or macrophyte samples, and in January the largest individuals from both sites were found in macrophytes. In 2017–18, the smallest individuals were never caught in macrophyte bait traps at either site and were more often found in bare habitats or CPOM/depositional area traps than snags. In December, the smallest animals from each site were found in bare habitats; in later months, the habitat the smallest individual was present in was not consistent between sites. In 2016–17, the smallest individuals were more likely to be caught in bait traps placed in bare or macrophyte habitats; given these habitats were also where the largest individuals tended to be caught, this implies bare and macrophyte habitats supported individuals of a variety of sizes. In terms of habitat use by month, the smallest individuals captured were consistently associated with particular habitats for each month (with the exception of February) (consistent = same habitat at both sites); in December, they were caught in bare habitats and CPOM/depositional areas, in January they were in macrophytes and in March they were in bare habitats.

The average size of *Paratya australiensis* caught in bait traps was larger than those caught in RESS samples and also tended to be less variable. In general, average size tended to decrease slightly at both sites in February and March compared to December and January in 2017–18 (Figure 5-11b). In contrast, average sizes did not vary much across the months in 2016–17 (although in most months, average sizes were smaller than equivalent months in 2017–18). Due to the low abundance and patchy distribution of *P. australiensis* in the bait traps, no patterns of habitat type and animal size were obvious.

Reproduction

As in 2016–17, no ovigerous female *Macrobrachium australiense* were caught in February or March (Figure 5-12a). Ovigerous females tended to be of moderate size, with carapace lengths between 16mm to 21mm, which was well below the average and maximum carapace lengths in those months. This result is also consistent with 2016–17, indicating that breeding is restricted to warmer months and that females must reach a minimum size before reproducing but are presumably smaller than males. Also of note is that in 2017–18, ovigerous females were never caught in macrophytes at either site and seemed to show a preference for bare habitat, especially at Loch Garry, which may have been a behaviour to avoid complex habitats (such as macrophytes) where larger dominant individuals would be present.

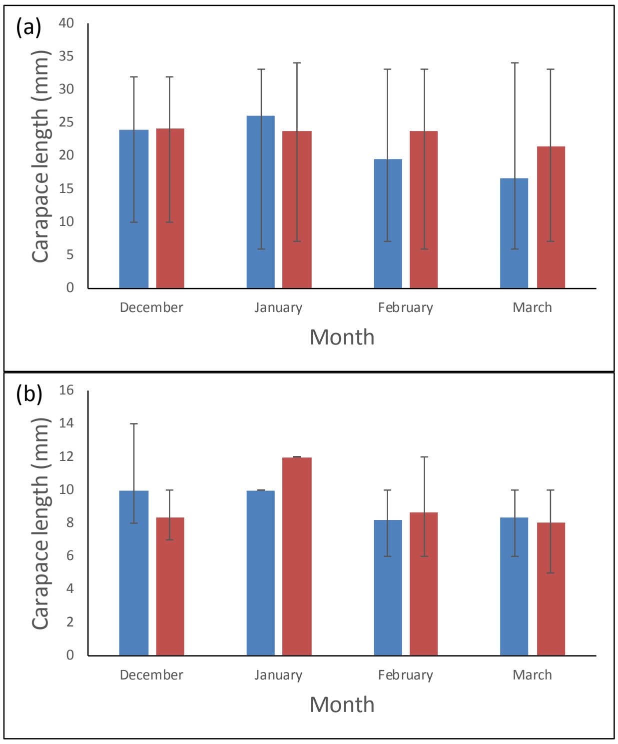


Figure 5-11. Average carapace lengths for (a) *Macrobrachium australiense* and (b) *Paratya australiensis* in bait traps from 2017–18. Error bars are the minimum and maximum carapace lengths, while blue columns = McCoys Bridge and red columns = Loch Garry.

The average percentage of ovigerous *P. australiensis* caught in bait traps was lower in 2017–18 than in 2016–17 (when numbers of *P. australiensis* were also lower). No ovigerous females were caught in March during both survey years (Figure 5-12b). Carapace lengths of ovigerous females ranged from 8mm to 12mm, similar to the previous year and similar to or greater than the average carapace length of all individuals caught in the bait traps. Like with *M. australiense*, this indicates breeding is limited to warmer months; however, ovigerous female *P. australiensis* are not considerably smaller than the largest individuals caught, showing a significant difference between the two crustacean species, where there appears to be very strong sexual dimorphism in *Macrobrachium* (with much larger males) that is not the case for *Paratya*. Another difference is that ovigerous *P. australiensis* showed a clear preference for bait traps placed in more complex habitats (snags and macrophytes), with only one caught in a depositional area, and this might reflect either a refuge seeking behaviour while ovigerous (to avoid predation) or trying to find more sheltered areas to release young. A similar result was observed in 2016–17.

5.5.4 Additional crustacean surveys: RESS

Abundance and biomass

As in the previous year, *P. australiensis* were the oust abundant crustaceans in RESS samples. At both Goulburn River sites abundances increased in the summer months before dropping off in March (Figure 5-13a). A comparison of abundances in RESS samples taken from all years at McCoys Bridge showed that the spring freshes delivered in 2015–16 and 2017–18 did not have as great a positive impact on *P. australiensis* abundances as the spring floods in 2016–17 (or the negative impact of the blackwater event later that year) (Figure 5-13b). However, the higher abundances pre- and post-CEW this year compared to 2015–16 do indicate that there may be some sort of positive, accumulative effect of CEW and floods on *P. australiensis* in the lower Goulburn River.

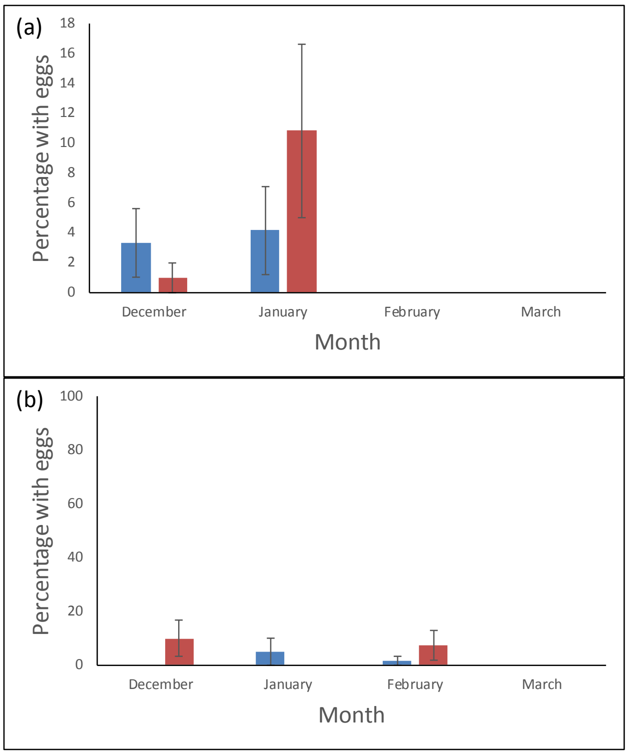


Figure 5-12. Percentage of (a) *Macrobrachium australiense* and (b) *Paratya australiensis* captured in bait traps in 2017–18 that were ovigerous (average ± standard error of the mean). Blue columns = McCoys Bridge and red columns = Loch Garry.

Paratya australiensis biomass did not follow abundance. It decreased in December at McCoys Bridge post-CEW and increased over the summer months and was much greater in February than was indicated by abundance, suggesting the population during this time was made up of larger individuals (rather than more individuals) (Figure 5-13c). In contrast, biomass was lowest at Loch Garry in January when abundance happened to be greatest, and at this time the population at Loch Garry was dominated by very small *P. australiensis* inhabiting recently inundated terrestrial vegetation (Figure 5-13c).A comparison of biomass across the years at McCoys Bridge again highlights the huge impact of natural spring floods and the blackwater event in 2016–17 on biomass compared to CEW in surrounding years (Figure 5-13d).

Macrobrachium australiense abundances decreased in edge samples following the spring fresh at both sites, before increasing over the remaining summer months (Figure 5-14a). The low abundance in December sharply contrasts with post-CEW and post-flood results from the same month in previous years at McCoys Bridge (Figure 5-14b), which would suggest that some other factor rather than the spring freshes caused this reduction. One possible factor could be low dissolved oxygen which was measured in the lower Goulburn River around this time.

Unlike abundances, which were similar between the two Goulburn sites over the months, *M. australiense* biomass differed between sites, indicating that the size of the animals in the population at each site was an important factor in determining biomass, as opposed to abundance alone (Figure 5-14c). As with abundance, a comparison of biomass across the years at McCoys Bridge did not show a consistent effect of spring freshes on biomass (again, perhaps due to low dissolved oxygen during post-CEW sampling in 2017–18). However, what was apparent was the massive increase in biomass during 2016–17 following natural floods in spring, with this increase persisting even into the time when the river was impacted by blackwater (Figure 5-14d).

In 2017–18, immature crustacean abundance was highest earlier in the sampling period, with the greatest abundance observed in December at McCoys Bridge and January at Loch Garry before dropping off in later months (Figure 5-15a). In contrast, in 2016–17 abundances were highest at both sites in January, and this persisted into February. Due to their small size, immature crustaceans contributed little to crustacean biomass even when they were highly abundant. Changes in their biomass over the months closely matched changes in their abundance (Figure 5-15b).

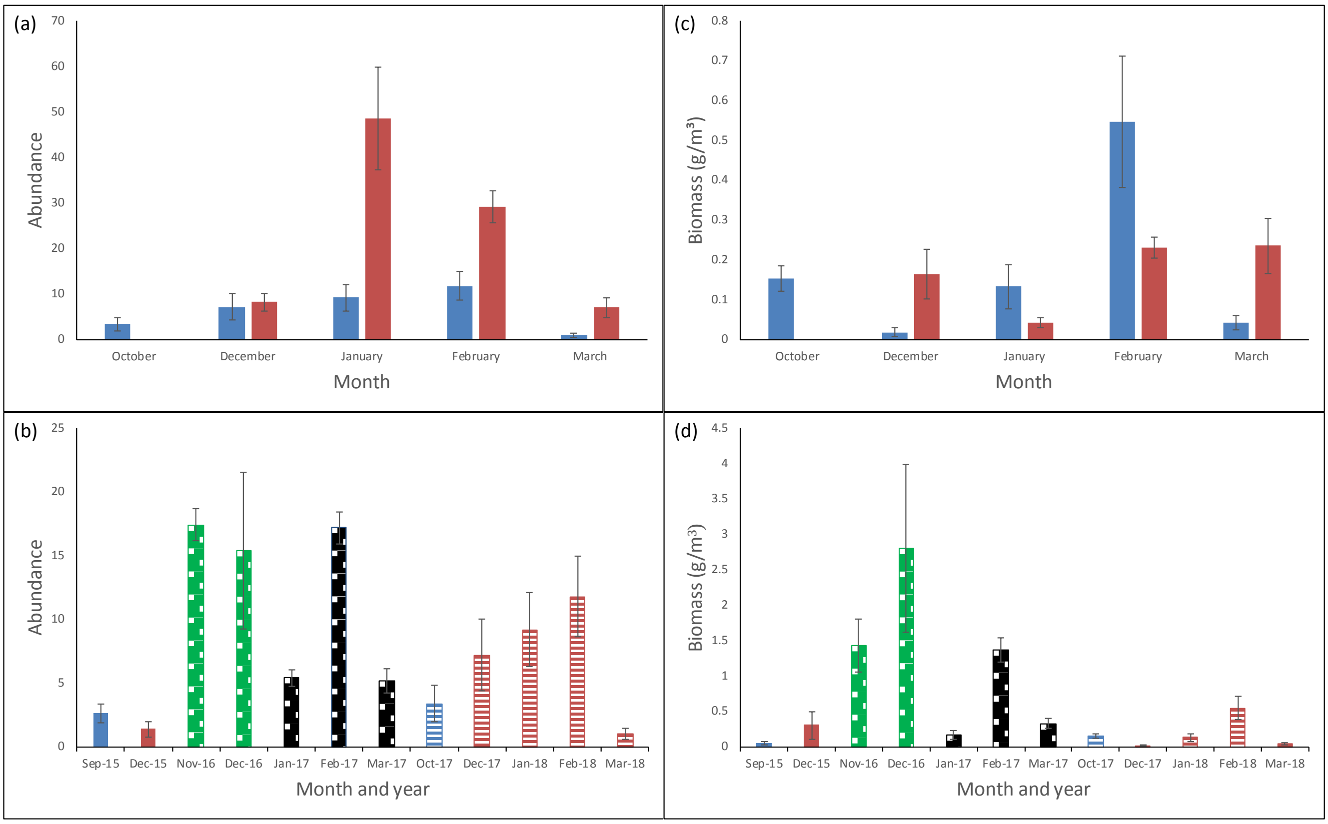


Figure 5-13. *Paratya australiensis* (a) abundance in 2017–18 at both sites, (b) abundance across all years at McCoys Bridge only, (c) biomass in 2017–18 at both sites and (d) biomass across all years at McCoys Bridge only. Values are average ± standard error of the mean. For figures (a) and (c), blue columns = McCoys Bridge and red columns = Loch Garry. For figures (b) and (d), blue = pre-CEW, red = post-CEW, green = post-natural flood (2016-17 only), black = post-blackwater event (2016–17 only); solid = 2015–16 sampling year, stippled = 2016–17 sampling year, striped = 2017–18 sampling year.

Carapace lengths

Paratya australiensis lengths were quite variable at each site and in each month (as seen with the error bars in Figure 5-16a), with the average length of animals indicating that while some large animals were present, most tended to be smaller. The high variability in sizes reflects the fact that the population at both sites consists of differently aged individuals. The average size of individuals tended to increase over the months at McCoys Bridge, whereas at Loch Garry the average size decreased in January before increasing again over the months. This differs from what was observed in 2016–17, where the average size decreased at McCoys Bridge after the blackwater event in late December then slowly increased over subsequent months, whereas the average size was fairly consistent from December to February at Loch Garry before declining in March.

Macrobrachium australiense collected from edge habitats were also highly variable in size, including very large (carapace length > 30mm) and very small (carapace length < 3mm) individual (Figure 5-16b). Again, the average size of animals caught at each site and in each month tended to be medium to smaller sized individuals (from 7 to 16mm carapace length). Average lengths were similar across sites and decreased in January. However, sites differed in their variability in *M. australiense* sizes, with animals caught at Loch Garry more similar in size than at McCoys Bridge. In contrast, the average carapace lengths of *M. australiense* caught in 2016–17 tended to be similar at both sites and were generally larger than those caught in 2017–18.

Reproduction

As in 2016–17, few ovigerous *P. australiensis* were caught in RESS samples, with the greatest percentage of ovigerous *P. australiensis* caught at Loch Garry in December (Figure 5-17a). Consistent with the previous year and the bait trap results, no ovigerous females were caught in March. Ovigerous female carapace lengths ranged from 9mm to 15mm, which is similar to 2016–17 (8mm to 17mm) and greater than average carapace lengths from the same months (if not equal to or similar to maximum carapace lengths). Again, this demonstrates mature female *P. australiensis* are not smaller than males.

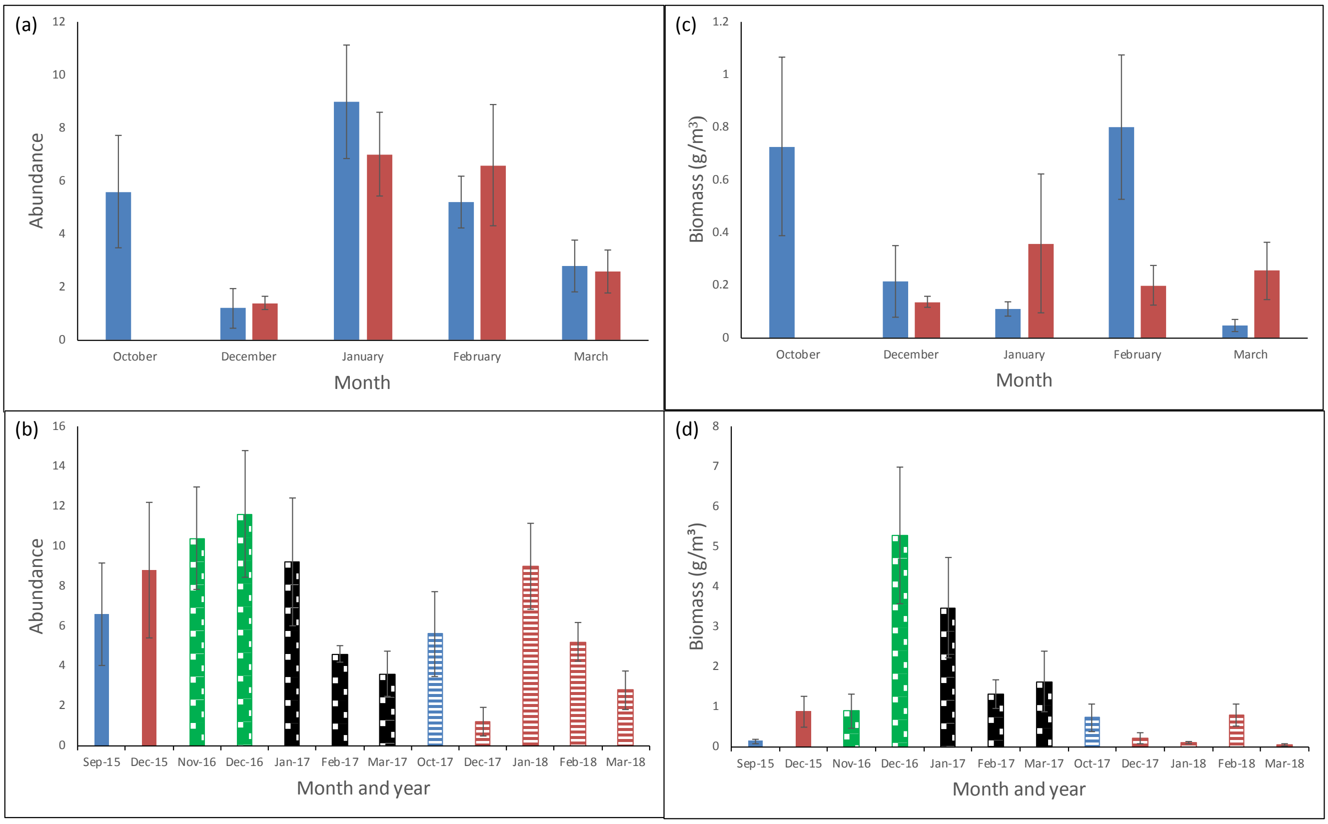


Figure 5-14. *Macrobrachium australiense* (a) abundance in 2017–18 at both sites, (b) abundance across all years at McCoys Bridge only, (c) biomass in 2017–18 at both sites and (d) biomass across all years at McCoys Bridge only. Values are average + standard error of the mean. For figures (a) and (c), blue columns = McCoys Bridge and red columns = Loch Garry. For figures (b) and (d), blue = pre-CEW, red = post-CEW, green = post-natural flood (2016–17 only), black = post-blackwater event (2016–17 only); solid = 2015–16 sampling year, stippled = 2016–17 sampling year, striped = 2017–18 sampling year.

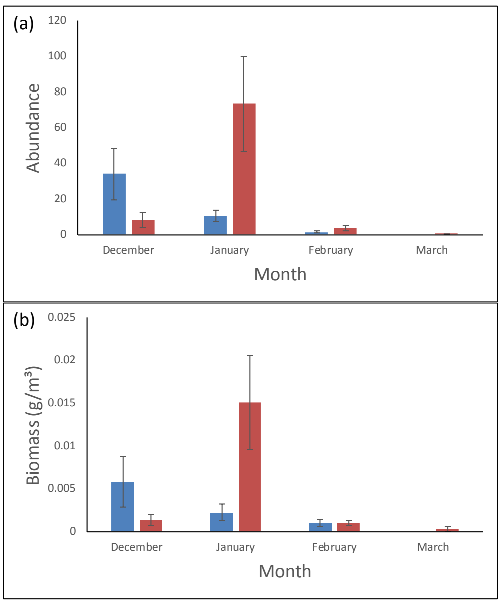


Figure 5-15. Immature crustacean (a) abundance and (b) biomass in RESS samples from 2017–18 (average + standard error of the mean. Blue columns = McCoys Bridge and red columns = Loch Garry.

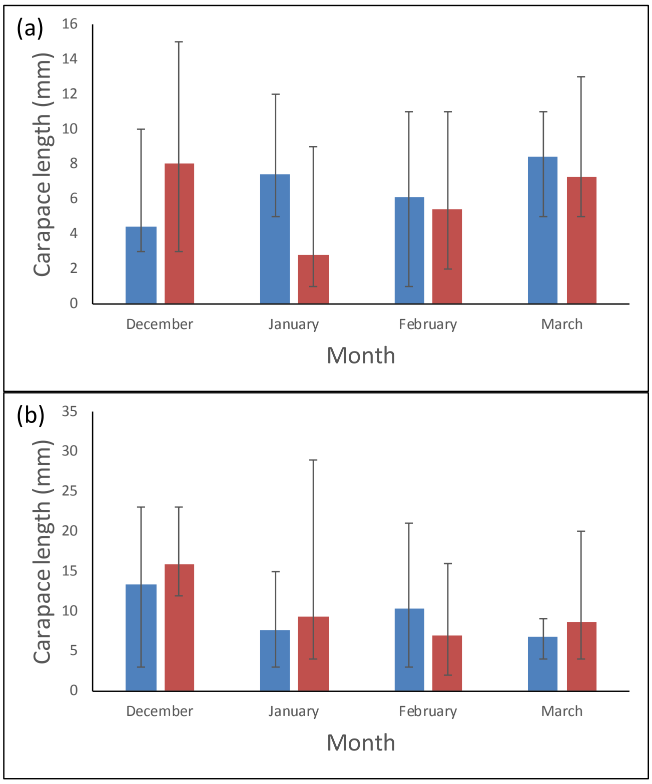


Figure 5-16. Average carapace lengths of (a) *Paratya australiensis* and (b) *Macrobrachium australiense* in RESS samples from 2017–18. Error bars = minimum and maximum carapace lengths. Blue columns = McCoys Bridge and red columns = Loch Garry.

In 2017–18, ovigerous *M. australiense* were only present in February (as was observed in bait traps) (Figure 5-17b); in contrast, they were present in both December and January only in 2016–17. Ovigerous female carapace lengths ranged from 8mm to 9mm, which is smaller than 2016-17 and in bait traps from 2017–18. Although these were similar to the average carapace lengths of *M. australiense* caught in February, they were still smaller than the maximum carapace lengths.

5.6 Discussion

The results from the 2017–18 survey period continue to support the notion that macroinvertebrates are responding to increased flows in spring. Responses are not observed across all taxa or across all endpoints, but the consistency in some responses do provide evidence that flows are having an impact, especially on crustacean biomass and abundance in the Goulburn River. However, it also needs to be noted that this evidence points to only a small impact of CEW when delivered as spring freshes; comparisons of data from years when spring freshes were delivered to the 2016–17 spring floods show that the much larger, natural flows achieved in that year had a greater positive impact on crustacean abundances and biomass, presumably due to organic matter entrainment into the river channel that cannot be achieved through spring freshes.

Spring freshes and other environmental water delivery do have smaller positive impacts on the macroinvertebrate fauna in the Goulburn River, especially through the inundation and maintenance of important habitats. Without environmental water delivery and modification of other flows from winter through to the end of summer, bank vegetation and aquatic vegetation growth and maintenance through drier months would be suppressed, with implications for aquatic invertebrates. These habitats provide an important source of food and shelter to macroinvertebrates. An excellent example of how important these habitats came from RESS samples taken at Loch Garry in January 2018. Here, a combination of earlier freshes along with elevated summer flows supported dense bank vegetation (including grasses) that were inundated by water during the elevated flows. Bank condition at this site is generally steep and undercut, but with the inundation of grasses a sheltered environment was present that was able to support numerous immature crustaceans that would otherwise have been washed downstream.

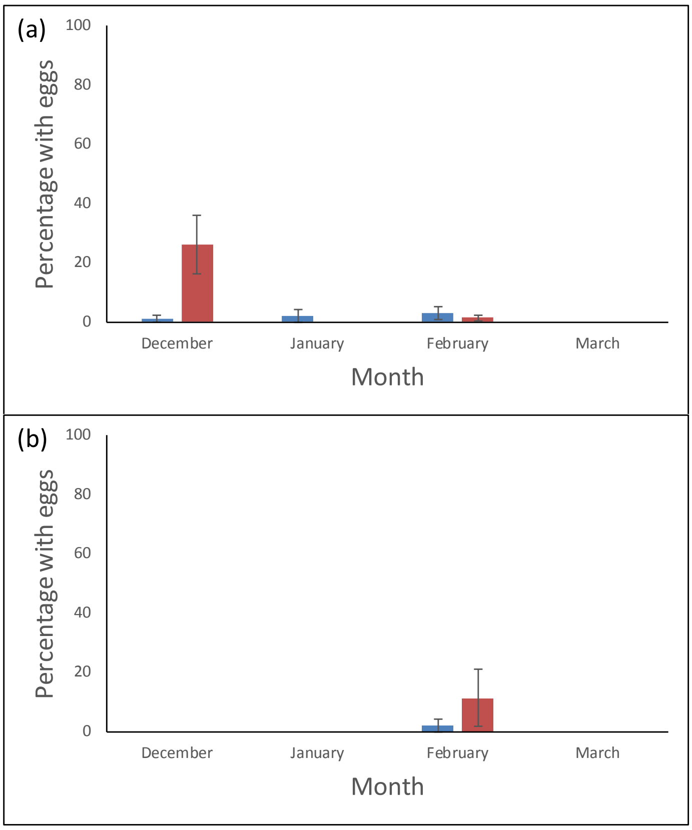


Figure 5-17. Percentage (a) *Paratya australiensis* and (b) *Macrobrachium australiense* in RESS samples in 2017–18 that were ovigerous (average ± standard error of the mean). Blue columns = McCoys Bridge and red columns = Loch Garry.

One difficulty that has previously been faced by macroinvertebrate monitoring is understanding the mechanisms behind how flows affect macroinvertebrates. The hypothesis that CEW spring freshes (and natural floods) affect macroinvertebrates through bottom-up effects (i.e. increased productivity) was not supported by analysis of the rates of primary productivity and respiration in earlier years of the Goulburn LTIM Project. However, consideration of metabolism in terms of the amount of carbon fixed, a new approach to analysis this year (Section 4), has provided convincing evidence of the effect of both managed and natural high flows on metabolic inputs to the food chain. To further address the links between stream metabolism and macroinvertebrates, additional monitoring has been commissioned for 2018–19 to examine how algal biofilms respond to different flows in spring and summer. Understanding how algal biofilm community structure and biomass change in response to something such as a spring fresh will help to further disentangle the metabolism signal, and could provide an answer about how these flows affect the macroinvertebrates that feed on biofilms. Questions have also been posed about how variable winter flows might benefit macroinvertebrates and how macroinvertebrate habitat use can change in the short term in response to changes in water heights. An intensive crustacean survey will be carried out during winter 2018 to answer these questions.

6. Vegetation Diversity

6.1 Introduction

Riparian and aquatic vegetation underpins aquatic systems by: (1) supplying energy to support food webs, (2) providing habitat and dispersal corridors for fauna, (3) reducing erosion and (4) enhancing water quality. In the Goulburn River drought and floods have reduced the quantity, quality and diversity of riparian and bankside vegetation over the last 20 years. Minimum summer and winter low flows and periodic freshes are recommended to help rehabilitate and maintain vegetation along the lower Goulburn River. The recommended flow components shape aquatic plant assemblages by influencing (1) inundation patterns in different elevation zones on the bank and hence which plants are promoted in each zone; (2) the abundance and diversity of plant propagules dispersing in water; and (3) where those propagules are deposited and germinate.

Vegetation diversity was monitored at four sites in the lower Goulburn River as part of the Victorian Environmental Flows Monitoring and Assessment Program (VEFMAP; Miller et al. 2015) and the Commonwealth Short Term Monitoring Projects (STIM; Stewardson et al. 2014, Webb et al. 2015). Vegetation diversity monitoring in the LTIM Project at two sites in the lower Goulburn River is extending those data sets and allowing the effect of different flow components to be assessed in wet and dry climatic conditions. The results are being used to identify what flows are needed to maintain or rehabilitate riparian vegetation in the lower Goulburn River depending on its current condition and state of recovery. They are also being used to broadly inform appropriate water management in other systems recovering from extreme events.

6.2 Area specific evaluation questions

To determine the contribution of Commonwealth environmental water in selected areas, and to improve understanding of the relationship between specific watering actions and ecological objectives for assets, the following questions are being addressed. This information also forms the basis of Basin-scale evaluation – where area-level results are scaled up to the Basin level.

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
What has CEW contributed to the recovery (measured through species richness, plant cover and recruitment) of riparian vegetation communities on the banks of the lower Goulburn River that have been impacted by drought and flood and how do those responses vary over time?	The spring fresh delivered in 2017 was appropriate but was followed by natural high flows between 19 November to 27 December 2017.	The effect of environmental flows in 2017-18 are obscured by the high natural flow event over November and December 2017. This natural event meant that vegetation on the bank face had a limited window for growth between successive inundation events. In addition, high natural flows in December 2017 restricted sampling. These factors have limited our ability to evaluate the effects of the spring fresh in 2017. The data however provides insights of responses to more prolonged inundation over spring-summer. Trends in cover at sampling locations show that more prolonged inundation over spring-summer constrained increases in cover usually observed between the pre and post fresh sampling. It is expected that plants would have increased in cover following the recession of high flows, but sampling does not cover this period.	Trends in cover of different taxa and groups of taxa over time and across the elevation gradient.
How do vegetation responses to CEW delivery vary between sites with different channel features and different bank conditions?		Reponses of vegetation to environmental water and unregulated flows are similar at McCoys Bridge and Loch Garry. However, vegetation	Trends in cover of different taxa and groups of taxa over time and

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
		cover is consistently lower at McCoys Bridge compared with Loch Garry.	across the elevation gradient
Does the CEW contribution to spring freshes and high flows trigger germination and new growth of native riparian vegetation on the banks of the lower Goulburn River?		The effect of the spring fresh in 2017–18 are obscured by the high natural flow event over November and December 2017. This natural event meant that vegetation on the bank face had a limited window for growth between successive inundation events. It is likely that following the recession of natural high flows in December 2017 that the cover of water dependent species would have increased but additional sampling would have been required to detect this.	Trends in cover
How does CEW delivered as low flows and freshes at other times of the year contribute to maintaining new growth and recruitment on the banks of the lower Goulburn River?		Natural high flows are likely to have exerted a strong influence on vegetation responses in 2017 and it is not possible to infer the influence of CEW delivered at other times of the year.	

6.3 Main findings from the vegetation monitoring program

6.3.1 Findings from 2017–18

- The effect of the spring fresh in 2017–18 was obscured by a natural high flow event over November and December 2017 which limited plant growth following the spring fresh. Sampling was also more limited as high flows had not fully receded when sampling was undertaken in December 2017.

- The data however reveal that the cover of *Poa labillardierei* (common tussock grass) which occupies higher elevations on the bank face tended to increase between September and December 2017. This may reflect the benefits of flows on improving soil moisture at higher elevations. In contrast, the cover of water dependent species that occupy lower elevations on the bank and experienced longer periods of deeper inundation declined slightly between sample events.

- Temporal patterns indicate that the ground layer vegetation cover sampled in September and December increased by ~ 20% between 2014 and 2017. Most of the observed increases in ground layer cover is due to increased cover of grasses, particularly *Poa labillardierei* (common tussock grass). In contrast to grasses, the cover of water dependent species as a group have oscillated and only a marginal increase in cover at Loch Garry only has been sustained in September and December 2017, and this increase is due to *Persicaria prostrata* (creeping knotweed).

6.3.2 How these build on findings from years 1 to 3

- The cover of water dependent vegetation across all sampling locations at both sites increased following spring freshes in 2014–15 and 2015–16 while the cover of all grasses decreased. While this pattern is correlated with spring freshes it is not known what portion of the increase in cover can be attributed to seasonal patterns of plant growth that would have occurred without the delivery of spring freshes. In 2017–18 natural high flows following the spring fresh extended inundation and cover of water dependent vegetation declined.

- The extent and duration of inundation provided by spring freshes is correlated with the distribution and cover of vegetation along the bank. Several plant species that have an affinity for wet habitats have higher cover in regions of the bank inundated by spring freshes. In contrast, the perennial native grass *Poa labillardierei* (common tussock grass) is more restricted in its distribution to elevations at or above the level inundated by spring freshes. This pattern of species distribution along the elevation profile has persisted in 2017–18.

- The recruitment of woody species, specifically silver wattle (*Acacia dealbata*) and river red gum (*Eucalyptus camaldulensis*) is generally restricted to higher areas of the bank which experience shallow and less frequent inundation. This pattern has persisted in 2017–18.

- Climatic conditions and non-regulated flows can exert a strong influence on vegetation and potentially influence the outcomes of environmental watering actions. Drier conditions in 2014–15 resulted in the recruitment of sedges along the river margin at base flow but a reduction in the cover and spatial extent of *Alternanthera denticulata* (lesser joyweed). In contrast, prolonged natural flooding in 2016–17 caused a substantial decline in the cover and occurrence of establishing sedges but increased the cover and distribution of *A. denticulata* and to a lesser extent *Centipeda cunninghamii* (common sneezeweed). The cover of Cyperaceae in 2017–18 remained low and natural high flows in 2017–18 did not greatly influence the cover of this group.

- Some species such as *A. denticulata* (lesser joyweed) and *C. cunninghamii* (common sneezeweed) can increase when exposed wet mud is available on the recession of high flows, and show a dynamic pattern of occurrence and cover both spatially and temporally. Other species such as *P. prostrata* maintain a more stable position along the elevation gradient possibly supported by a persistent woody root stock. Data collected in 2017–18 has not revealed any major change in the cover or spatial distribution of the taxa examined.

- There was no evidence that the delivery of a fresh delivered in March 2017 had any immediate negative outcome on bankside vegetation. There is some evidence that grasses benefited from this late season watering. No data were collected in 2017–18 to evaluate the influence of flows at other times of the year.

- Modelled relationships between the cover of selected taxa and duration of inundation the year prior to sampling reveal that the hydrologic envelopes differ for various groups and species examined. The data collected in 2017 has contributed to refining these models.

- Changes in the cover of examined taxa over time are similar at Loch Garry and McCoys Bridge but the cover of all taxa examined was lower at McCoys and the gradual increase in cover of *P. prostrata* over time observed at Loch Garry was not evident at McCoys Bridge. These patterns persisted in 2017–18.

- The reason for differences in cover at the two sites is not known but may reflect differences in channel shape, aspect of sampled transects, or differences in subsurface water inflows. Loch Garry potentially receives higher subsurface water inflows from the closer proximity of large wetlands compared to McCoys which experiences more human activity and goat grazing on *P. prostrata* (*pers*. *obs.* D. Lovell, GBCMA).

6.4 Methods

6.4.1 Sampling

Elevation surveys

Vegetation responses to flow are expected to vary with elevation as this determines the depth and duration of inundation experienced under a particular flow. To support more targeted monitoring, elevation profiles were obtained at 1 m intervals along all transects in December 2014 using a high-precision RTK GPS. These were used to target sampling locations along each transect in 2015–16 to ensure an optimal range of elevations was sampled along each transect.

Elevation profiles were surveyed again in December 2016 following the recession of floodwater to ensure accurate inundation histories of sampling locations. Elevation surveys in December were supported by the GBCMA with funding from VEFMAP.

Vegetation sampling

Vegetation was sampled on both banks at Loch Garry and McCoy's Bridge, before and after the delivery of spring freshes in 2014–15, 2015–16 and 2017–18 (Table 6-1). In 2016 spring freshes were not delivered due to the large natural high flows that persisted between June and November 2016, and vegetation was instead sampled in December 2016 after the recession of flood waters. Comparing vegetation cover measured in December 2016 with past surveys in December 2014 and 2015 provides insights into the influence of large natural flood events.

Table 6-1. Summary of vegetation survey dates, sampling locations and transects.

Year	Survey Number	Survey Type	Date	Sites sampled	Transects sampled North bank	Transects sampled South bank
2014-15	1	Pre spring fresh	23 Sept & 3 Oct 2014	Loch Garry	1,3,5,8,9,10,12,13,15	9,10,11,12,13
			24 Sept 2014	McCoy's Bridge	1,2,3,6,8,10,12,13,15	1,2,3,5,10,12,13,15
	2	Post spring fresh	16 Dec 2014	Loch Garry	1,3,5,8,9,12,13,15	1,3,5,9,10,12,13,15
			17 Dec 2014	McCoy's Bridge	1,2,3,6,10,12,13,15	1,2,3,6,10,12,13,15
2015-16	3	Pre spring fresh	16 Sept 2015	Loch Garry	1, 3, 5, 8, 9,10,12,13	1, 3, 5, 8, 9,12,13,15
			15 Sept 2015	McCoy's Bridge	1, 2, 6, 10, 12, 13,15	2, 3, 6,10,12,13,15
	4	Post-fresh	16 Dec 2015	Loch Garry	1, 3, 5, 8, 9,10,12,13	1, 3, 5, 8, 9,12, 13,15
			17 Dec 2015	McCoy's Bridge	1, 2, 3, 6,10,12,13,15	1, 2, 3, 6, 10, 12, 13, 15
2016-17	5	Post natural flood	12 Dec 2016	Loch Garry	1, 3, 5, 8, 9,10,12,13	1, 3, 5, 8, 9,12, 13,15
			13 Dec 2016	McCoy's Bridge	1, 2, 3, 6,10,12,13,15	1, 2, 3, 6, 10, 12, 13, 15
	6	Pre autumn fresh	21 Feb 2017	Loch Garry	1, 3, 5, 8, 9,10,12,13	1, 3, 5, 8, 9,12, 13,15
			22 Feb 2017	McCoy's Bridge	1, 2, 3, 6,10,12,13,15	1, 2, 3, 6, 10, 12, 13, 15
	7	Post autumn fresh	11 April 2017	Loch Garry	1, 3, 5, 8, 9,10,12,13	1, 3, 5, 9,12, 13,15
			10 April 2017	McCoy's Bridge	1, 2, 3, 6,10,12,13,15	1, 2, 3, 6, 10, 12, 13, 15
2017-18	8	Pre spring fresh	7 Sept 2017	Loch Garry	1, 3, 5, 8, 10,12,13	1, 3, 5, 8, 9,12,13,15
			8 Sept 2017	McCoy's Bridge	1, 2, 3, 6, 10, 12, 13,15	1, 2, 3, 6,10,12,13,15
	9	Post spring fresh	14 Dec 2017	Loch Garry	8, 9,10,12, 13	1, 3, 5, 8, 9,12, 13,15
			15 Dec 2017	McCoy's Bridge	1, 2, 3, 6,10,12,13,15	1, 2, 3, 6, 10, 12, 13, 15

Vegetation was again sampled in February 2017 and April 2017, before and immediately after, a fresh delivered in March 2017 for instream vegetation and fish objectives. Vegetation monitoring was undertaken in this case to assess recovery of vegetation following the natural flooding and to assess responses of vegetation to the March fresh that could guide future flow planning. Vegetation sampling carried out in April 2017 was supported by the GBCMA with VEFMAP funds.

At all sampling times vegetation was surveyed along transects that ran perpendicular to stream flow. Sampling was initially designed to survey regions of the bank that had previously been surveyed by other programs (i.e. VEFMAP and CEWO STIM). However, many quadrats sampled by these programs were at elevations well above the level expected to be inundated by spring freshes. As such, subsequent sampling did not attempt to match the spatial extent of these previous programs. Instead, surveys extended from around base flow to just above the level inundated by spring freshes (nominally a change in elevation of approximately 3 m). As transect elevation data were not available in the first year of sampling, a 3 m change in height from base flow was estimated visually.

At each sampling location 20 points were surveyed along a horizontal transect to give estimates of cover for each species (see details in standard operating procedures; Webb et al. 2018). Vegetation indicators were assessed using the line point intercept method at each sampling interval along the transect. This is done by placing a 2 m measuring tape perpendicular to the transect (i.e. parallel to streamflow) and recording every 10 cm along the tape all species that intercept a rod placed vertically through the vegetation. This gives a total of 20 sampling points at each sampling location. Foliage projected cover (%) for each species was then calculated by dividing the number hits per species by the total number of points sampled.

6.4.2 Analyses

Monitoring data collected over the four years of the LITM program provides insights into the responses of vegetation to environmental flow events and to longer term hydrologic regimes. Qualitative and quantitative approaches have been applied to evaluate vegetation responses.

Qualitative approaches include the following:

- Examination of percent foliage projective cover (FPC %) of different taxa across all sampled locations at each site in relation to short and longer-term flow histories.

- Examination of the foliage projective cover (FPC %) of different taxa across the elevation gradient at each sample date at each site.

Quantitative approaches were developed to (*i*) evaluate responses of vegetation to the March fresh and (*ii*) develop relationships between hydrologic variables and vegetation cover and occurrence that is more transferrable to other sites and support a more predictive approach.

The evaluation has concentrated on a subset of species with sufficient occurrences to reveal responses to inundation. More specifically, *Persicaria* spp., *Alternanthera denticulata* and *Poa labillardierei* are representative of ground-layer dominants of some Riverine floodplain Ecological Vegetation Classes (EVCs) relevant to the Goulburn River bankside assemblage (Cottingham et al. 2013). *Cyperus eragrostis* was included even though it is an introduced species, as it is representative of key ground-layer dominants of EVC 962 (Riparian Wetland), which develops in a band along the lower banks. The group "all grasses" included all annual and perennial, native and introduced grasses, but only *Poa labillardierei* occurred with high enough frequency to warrant species level analyses. *Water dependent* species were classified as those tolerant of flooding (Leck and Brock 2000).

Statistical Models: Relationships between hydraulic variables and vegetation

The data collected so far by the LTIM program represents an array of inundation histories at each sampling location generated by: (*i*) the range of elevation profiles sampled and (*ii*) differences in river discharge prior to vegetation sampling. A range of hydrological variables can be derived for each sampling time and location and used to characterise the hydrological envelope of vegetation.

Using the data collected by the LTIM program relationships between the total number of days inundated in the year prior to sampling and (*i*) vegetation abundance (% FPC) and (*ii*) the probability of occurrence of selected species/groups was examined.

The models described below for both vegetation presence and abundance were implemented in OpenBUGS version 3.2.1 (Lunn et al. 2009), using the R2OpenBUGS package (Sturz et al. 2005) in R (R Development Core Team 2010). Three independent Markov chains were used to confirm convergence of chains during model burn-in. Different burn-in periods were employed for different models, with the criterion for establishing convergence being an Rhat value of approximately 1 (Sturz et al. 2005). Different periods were also used for parameter estimation, based upon autocorrelation within the Markov chains.

i) Model of vegetation presence/absence and number of days inundated

Vegetation presence/absence (y_i) and was modelled as a non-monotonic function of flow within a Bayesian framework. The model is structured as follows:

$$y_i \sim Bernoulli(p_i) \quad \text{Equation 1}$$

$$logit(p_i) = int + eff.Q \times ({Q_i}^{\alpha} - 1)/\alpha + eff.Q2 \times [({Q_i}^{\alpha} - 1)/\alpha]^2 + \ eff.Transect_j \quad \text{Equation 2}$$

The presence/absence of vegetation species or groupings for site i has a Bernoulli distribution with a probability of p_i. p_i is modelled using a non-monotonic function and is driven by the global intercept (*int*), and the number of days that the sampling site is inundated in the previous year (Q_i), with α determining the shape of the function. In addition, there is a random effect of the transect in which the sampling site is located.

ii) Model of vegetation abundance and number of days inundated

When modelling vegetation abundance as a function of Q, y_i represents the cover (FPC) and is drawn from a Poisson distribution with an expected value of mu_i. mu_i is modelled using the same non-monotonic function as above.

$$y_i \sim Poisson(mu_i) \quad \text{Equation 3}$$

$$mu_i = int + eff.Q \times ({Q_i}^{\alpha} - 1)/\alpha + eff.Q2 \times [({Q_i}^{\alpha} - 1)/\alpha]^2 + \ eff.Transect_j \quad \text{Equation 4}$$

Both models were developed for grouped and individual vegetation species including:

- all species
- all water dependent taxa
- all grasses
- all introduced grasses
- all native grasses
- *Persicaria prostrata* (creeping knotweed)
- *Alternanthera denticulata* (lesser joyweed)
- *Poa labillardierei* (common tussock grass)
- Juncus spp.
- Cyperaceae

In each of models above, we also tested the impacts of rainfall over growing season for each year with a random effect of year (*eff_Year$_y$*), conditioned on growing season rainfall (*Rain$_{y_growing}$*), as:

$$eff_{Year_y} \sim Normal\left(mu_{eff_{Year_y}}, \sigma_{eff_{Year_y}}\right) \quad \text{Equation 5}$$

$$mu\_eff\_Year_y = int_{eff_{Year}} + eff_{Rain} \times Rain_{y\_growing} \quad \text{Equation 6}$$

The rainfall data for both sites were obtained at BoM rainfall station Kyabram, summed for each year over the previous growing season only (September to March).

We tested the above models for all species and grouped water dependent species with the LTIM monitoring data only (2014–2017).

An alternative model also compares the probability of occurrence of different vegetation types across the elevation profile at each site with a Bayesian logistic regression, to compare vegetation occurrence pre- and post-spring fresh. To eliminate effects of the flood in 2016, this analysis focuses on 2014, 2015 and 2017 data only.

6.5 Results

6.5.1 Relevant flow components delivered to the lower Goulburn River in 2017–18

2017–18: Spring fresh: Commonwealth environmental water was delivered to the Goulburn River for vegetation objectives over approximately 3 weeks, commencing the 20 September and finishing on the 9 October in accordance with seasonal watering plans (Figure 6-1). A maximum discharge of ~6680 ML/day was released. There were no further releases to meet fish objectives. Following the Spring fresh natural high flows occurred between 20 November 2017, reaching ~15558 ML/day at McCoys Bridge on 7 December 2017 and then falling to 976 ML/day on 28 December 2017. This natural event meant that higher than usual flows (~ 2300 ML/day) were experienced during vegetation surveys on 14-15 December 2017. Consequently, some transects could not be surveyed and the lowest elevation of all transects were inundated and could not be assessed.

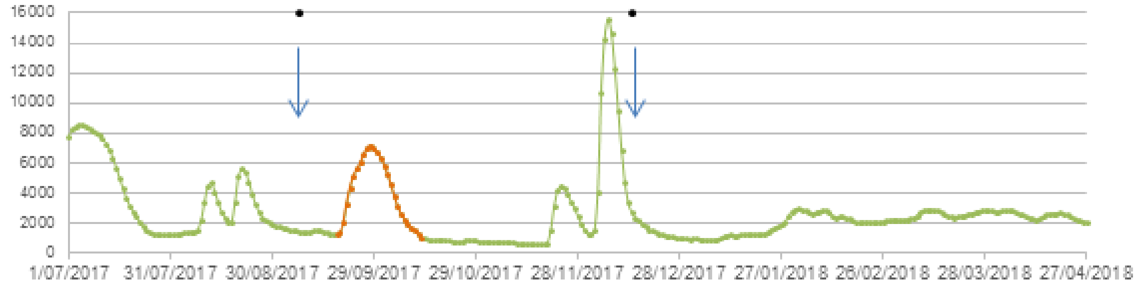


Figure 6-1. Goulburn river discharge (ML/day) for McCoy's Bridge in 2017—18 showing the spring freshes (orange). Blue arrows indicate timing of vegetation sampling.

6.5.2 Vegetation trajectories and flow 2017–18

Changes in mean cover over time at Loch Garry and McCoys Bridge are shown for different plants groups and species in Figure 6-2. As not all locations were accessible at each sampling event, only locations sampled at least 8 of the 9 sampling events were included to reduce bias in estimates resulting from differences in sampling at each time.

Temporal patterns in mean cover indicate that the cover of ground layer vegetation at sampled locations in September and December has increased by ~ 20% overall between 2014 and 2017. Most of the observed increases in total ground layer cover is due to an increased cover of grasses, particularly *P. labillardieri*. In contrast to grasses, the cover of water dependent species as a group have oscillated and only a marginal increase in cover at Loch Garry only has been sustained in September and December 2017, due to *P. prostrata.*

The mean cover of all grasses decreased between September and December in both 2014 and 2015, suggesting that spring freshes limit the cover of introduced grasses. In contrast, *P. labillardeirei* increased after spring freshes and reflects the higher elevations that this species occupies on the bank where spring freshes result in shorter and shallower inundation but increase soil moisture.

It was not possible to sample vegetation in September 2016 due to natural flooding, but the cover of grasses in December 2016 was similar to that recorded in December 2014 and 2015. This suggests that natural flooding may have produced a similar suppression of grass cover over spring and early summer.

Vegetation sampling in February and April 2017, before and immediately after the March 2017 fresh, found that the mean cover of all grasses increased over time, despite the fresh delivered in March 2017. This suggests that freshes later in the growing season when grasses are more mature may favour their growth.

The mean total cover of all water dependent species increased between September and December in 2014 and 2015, suggesting that spring freshes contributed to increasing cover. Increased covers following spring freshes in 2014 were not maintained and returned to similar levels in spring 2015. This may partly be attributed to a drier year. In December 2016 following natural flooding the cover of water dependent species was similar to that measured in December 2015. Water dependent vegetation increased in cover between February and April 2017 following natural flooding but returned to similar levels of cover in September and December 2017. It is unclear if the increased cover in February and April was due to the natural flood or if it represents a seasonal pattern of growth.

6.5.3 Vegetation responses to hydrologic conditions

Water dependent species differ in their hydrologic preferences. Patterns of mean cover for several water dependent taxa are shown in Figure 6-3 and reveal that the cover of *P. prostrata*, *A. denticulata* and sedges (Cyperaceae) all increased between September and December in 2014 and 2015 following spring freshes. This suggests that spring freshes maybe contributing to increased cover of these taxa, however it is uncertain how much change is due to seasonal patterns of plant growth that would occur in the absence of spring freshes.

Climatic conditions and unregulated river flows also influence vegetation and can override responses to environmental watering. In 2014–15 dry climatic condition and low unregulated flows over the year prior to monitoring in September 2015 was associated with reduced cover of *A. denticulata* while flooding in 2016–17 was associated with increased cover. In contrast, mean cover of sedges (Cyperaceae) did not decline over dry conditions in 2015–16 but was severely reduced in response to the prolonged flooding in 2016.

The cover of *P. prostrata* appears more resilient to variations in flow and climate conditions, particularly at Loch Garry where it has slowly increased in cover over time. A similar steady increase in cover of creeping knotweed has not been observed at McCoys Bridge. Similarly, the mean cover of all taxa examined was consistently lower at McCoy’s Bridge compared with Loch Garry.

6.5.4 Changes in patterns of species distribution along the elevation gradient

Species are not evenly distributed on the bank face but occur in zones that reflect each species tolerances to and affinity for the hydrologic regimes experienced at different elevations (Figure 6-4 and Figure 6-5). During periods where inundation experienced at an elevation on the bank is not favourable, the occurrence and or cover of the species may decline at that location but be maintained or increased at other locations on the bank that experience more suitable inundation regimes. Characterising the inundation regime at different elevations along the bank face over time provides insights into the hydrological envelope of each species.

The cover and distribution of native grasses and water dependent species along the elevation gradient show contrasting patterns with the cover of native grasses (mostly *P. labillardierei*), increasing at higher elevation while the cover of all water dependent species combined decreases at higher elevations (Figure 6-4). These patterns are similar at both Loch Garry and McCoys Bridge. The water dependent taxa examined also differ in their patterns of distribution along the bank face (Figure 6-5). *P. labillardierei* occupies the highest elevations sampled on the bank face and achieved highest cover at elevations above the level typically reached by spring freshes. In contrast *P. prostrata* occurs across a wide range of elevations but has the highest cover at mid elevations with cover declining above elevations typically reached by spring freshes and *A. denticulata* and Cyperaceae occupy comparatively lower elevations where inundation is more frequent indicating a greater dependence on water availability. As lower elevations are subject to the most pronounced variations in inundation depth and duration this likely contributes to the high variation in cover observed overtime for water dependent species.

The cover of some species along the elevation profile is dynamic and shifts over time. The distribution of *A. denticulata* shifted to lower elevations during drier condition in 2014–15 but increased again after the recession of flood water in 2016–17 (see previous reports). In contrast, the occurrence and cover of *Cyperus spp*. increased at lower elevations during the drier conditions in 2014–15 but decreased following prolong flooding in 2016–17). The distribution of *P. prostrata* along the elevation profile has not changed substantially over time.

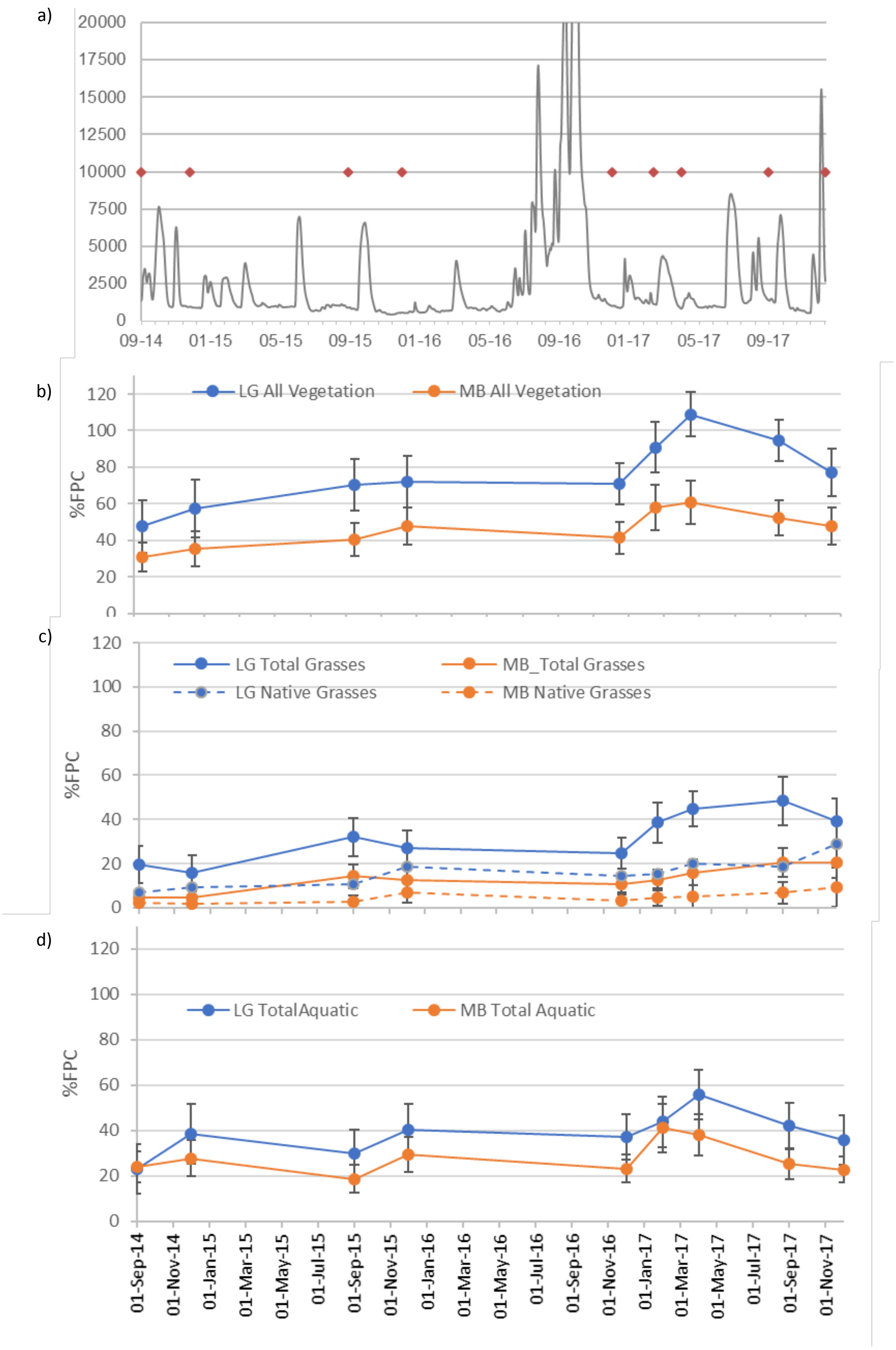


Figure 6-2. River discharge (Ml/day) over time (a). Mean foliage projected cover (FPC, %) (± 95% Confidence Intervals) at over time for all ground layer vegetation (b), total grasses and native grasses (c) and total water dependent species (d). Orange diamonds in represent the timing of vegetation surveys. Abbreviations: LG = Loch Garry, MB = McCoy's Bridge.

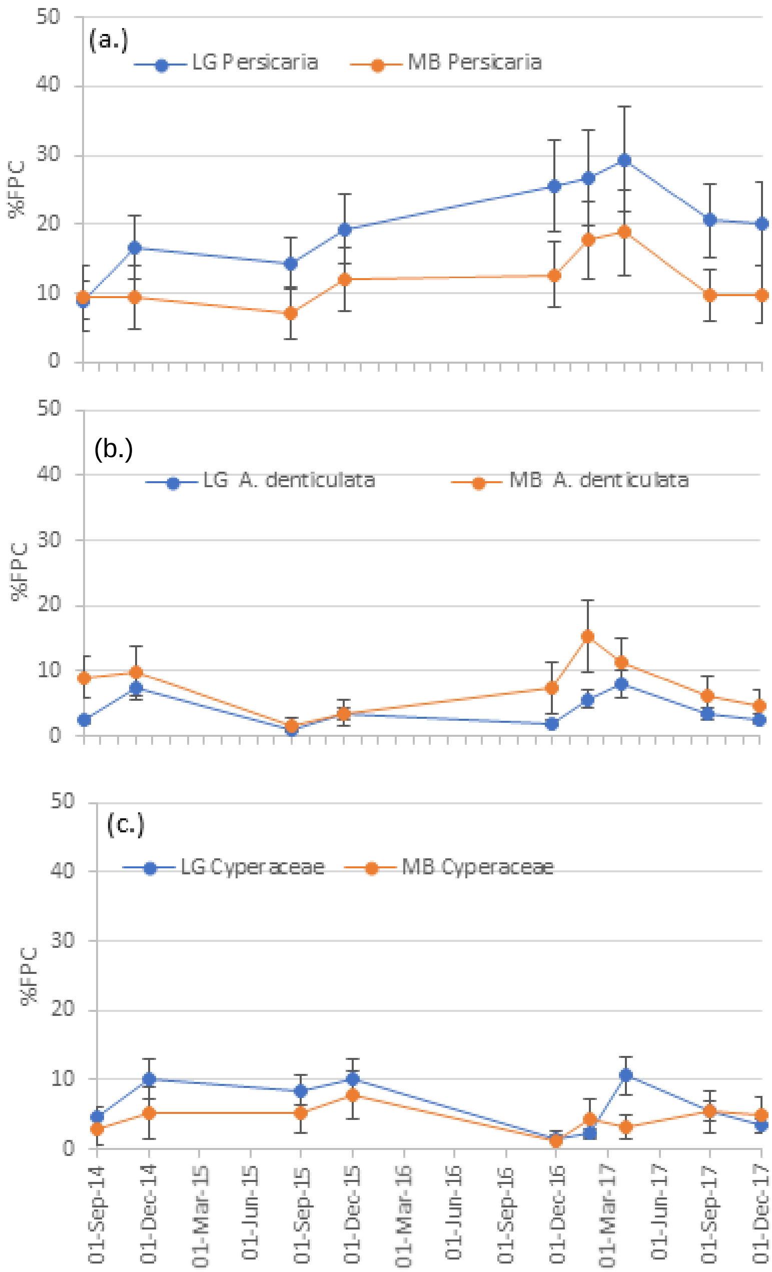


Figure 6-3. Mean FPC (%) (± 95% Confidence Interval) across all sampling location at Loch Garry and McCoy's Bridge at each sample date for *Alternanthera denticulata* (lesser joyweed), (a), *Persicaria prostrata* (creeping knotweed) (b) (middle panel), and Cypercaeae (sedges) (c). Abbreviations: LG = Loch Garry, MB = McCoy's Bridge.

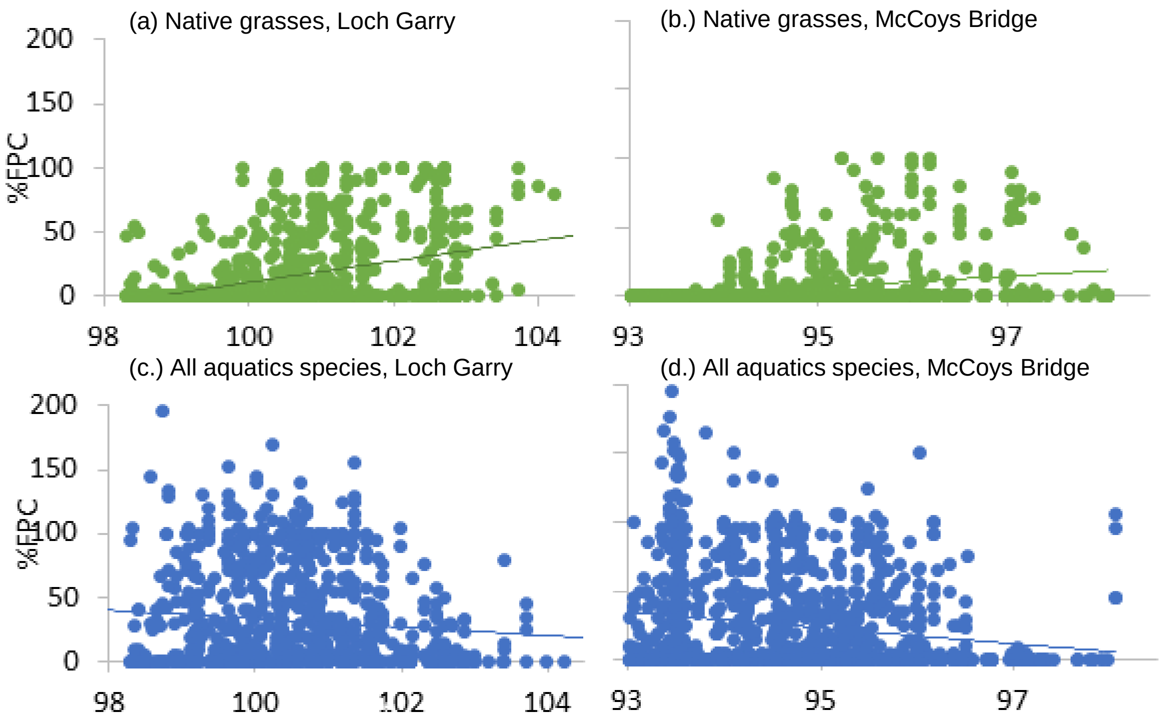


Figure 6-4. FPC (%) of native grasses (a, b) and all water dependent species (c, d) across the elevation gradient at Loch Garry (a, c) and McCoy's Bridge (b, d). Lines are logarithmic regressions between cover and elevation are shown.

6.5.5 Modelled responses of vegetation to hydrologic variables

Pattern of change in vegetation cover with number of days inundated in the year prior to sampling for all species and water dependent vegetation is shown for years 1 to 4. (Figure 6-6). Inclusion of rainfall over the growing season in the models (Equations 5 and 6) had a relatively minor role on the relationships between vegetation cover and inundation and were not considered in further modelling.

Modelled relationships of cover and inundation duration show similar response to increasing inundation duration each year but show that cover has increased between Year 1 and Year 4 regardless of inundation. Observed increases in cover may represent increased plant growth over time but cover estimates are also likely to have been inflated by differences in sampling season in Year 3 and Year 4.

Modelled patterns of cover for different plant group or species in response to days inundated the prior year using data from all nine sampling events between September 2014 and December 2017 are shown in Figure 6-7. The model outputs show that cover of all species as a group initially declines sharply with increasing inundation followed by a more gradual decline with increasing inundation. However, responses to inundation duration differ among the taxa examined. As such the value of models that include of all species combined has more limited value for management as it represent patterns of the most abundant group (Figure 6-7).

There was generally no significant difference between the probabilities of vegetation presence pre- and post-fresh in 2014, 2015 and 2017 combined (Figure 6-9). However, the natural flood that occurred following the spring fresh in 2017 probably limits the ability to detect responses. Future analyses should consider excluding events where natural flooding has occurred between the pre and post fresh surveys.

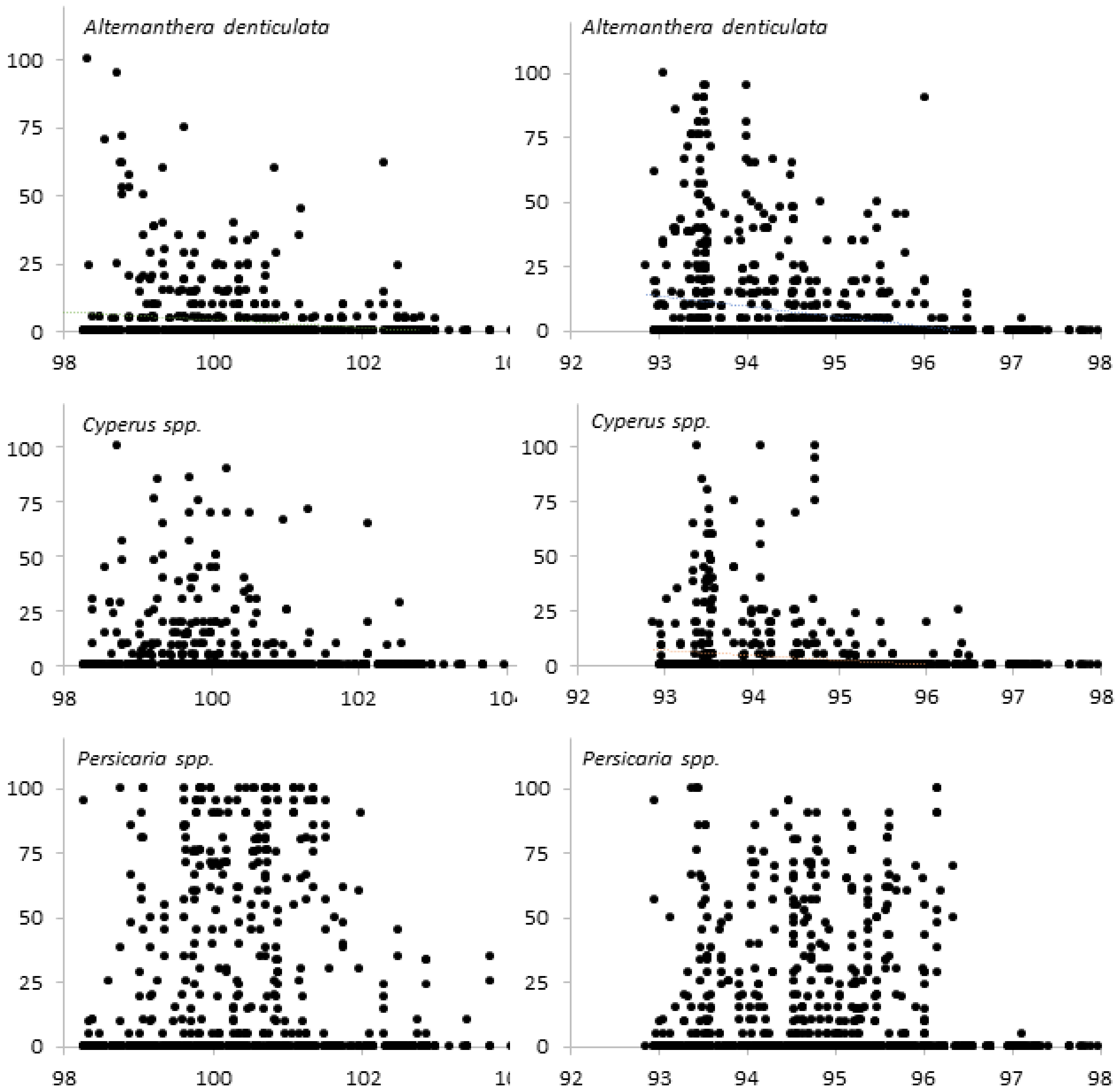


Figure 6-5. FPC (%)across *Alternanthera denticulata (*lesser joyweed), (upper panel), *Cyperus* species (middle panel) and *Persicaria prostrata* (creeping knotweed) (lower panel), at Loch Garry (left panel) and McCoy's Bridge (right panel).

6.6 Discussion

6.6.1 Issues

Natural high flows in November to December 2017 limited our ability to evaluate the response to spring freshes. Despite this data collected has provided insights into the consequences of more prolonged inundation over this period. However additional monitoring would have been valuable in understanding how vegetation responded following the recession of natural high flows.

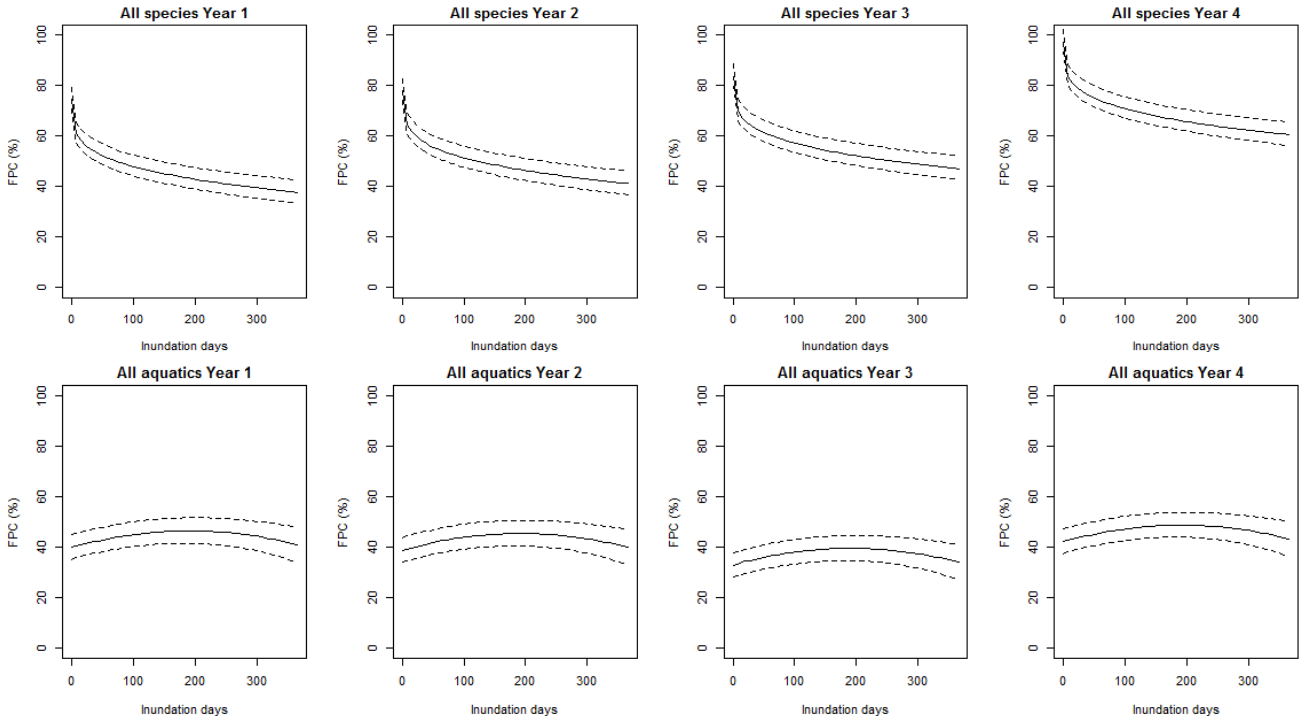


Figure 6-6. Modelled probability of foliage projected cover (FPC %) for all water dependent taxa (top row) and all species (bottom row), in response to number of inundation days in the previous year. Models also include the influence of rainfall over the growth period.

There would be value in understanding trajectories of plant growth over the growing season and how this is influenced by the timing of flow events. This would require additional surveys over the growing season but could be undertaken at one site and/or at a subset of transects at both sites.

6.6.2 Future analysis

- Future analyses examining the changes in the probably of occurrence of different plant groups or species pre-and post-freshes should exclude events where natural flooding has occurred between surveys.
- Trends in the cover of different groups and species should be analysed to determine if patterns are statistically significant.
- The influence of inundation depth and duration should be examined for lower elevations on the bank face where inundation depth is not expected to be strongly correlated with duration of inundation as it is at higher elevations.
- Temporal patterns of plant cover suggest that cover may increase into autumn. As such, models representing the relationships between inundation duration and vegetation cover should either include season as a term in the model or exclude data from different seasons (i.e. February and April surveys in 2016)

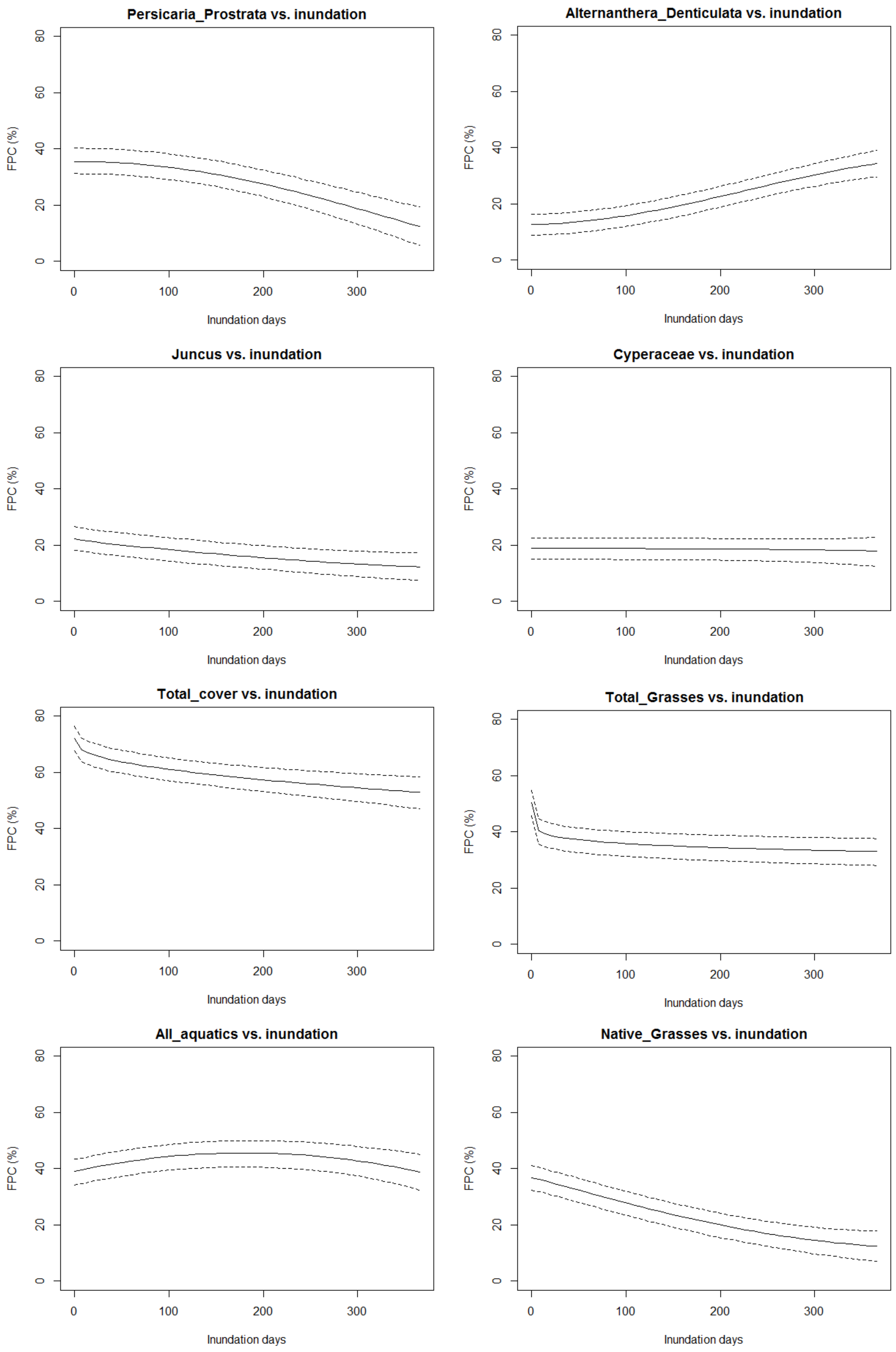


Figure 6-7. Modelled foliage projected cover (FPC %) for all different plant groups or species in response to number of inundation days in the previous year.

Modelled probabilities of occurrence in general do not show clear relationships with inundation and have high uncertainty for longer inundation (Figure 6-8). One exception is for the species *P. prostrata*, which has clear declining relationship when inundation increases, even with model uncertainty.

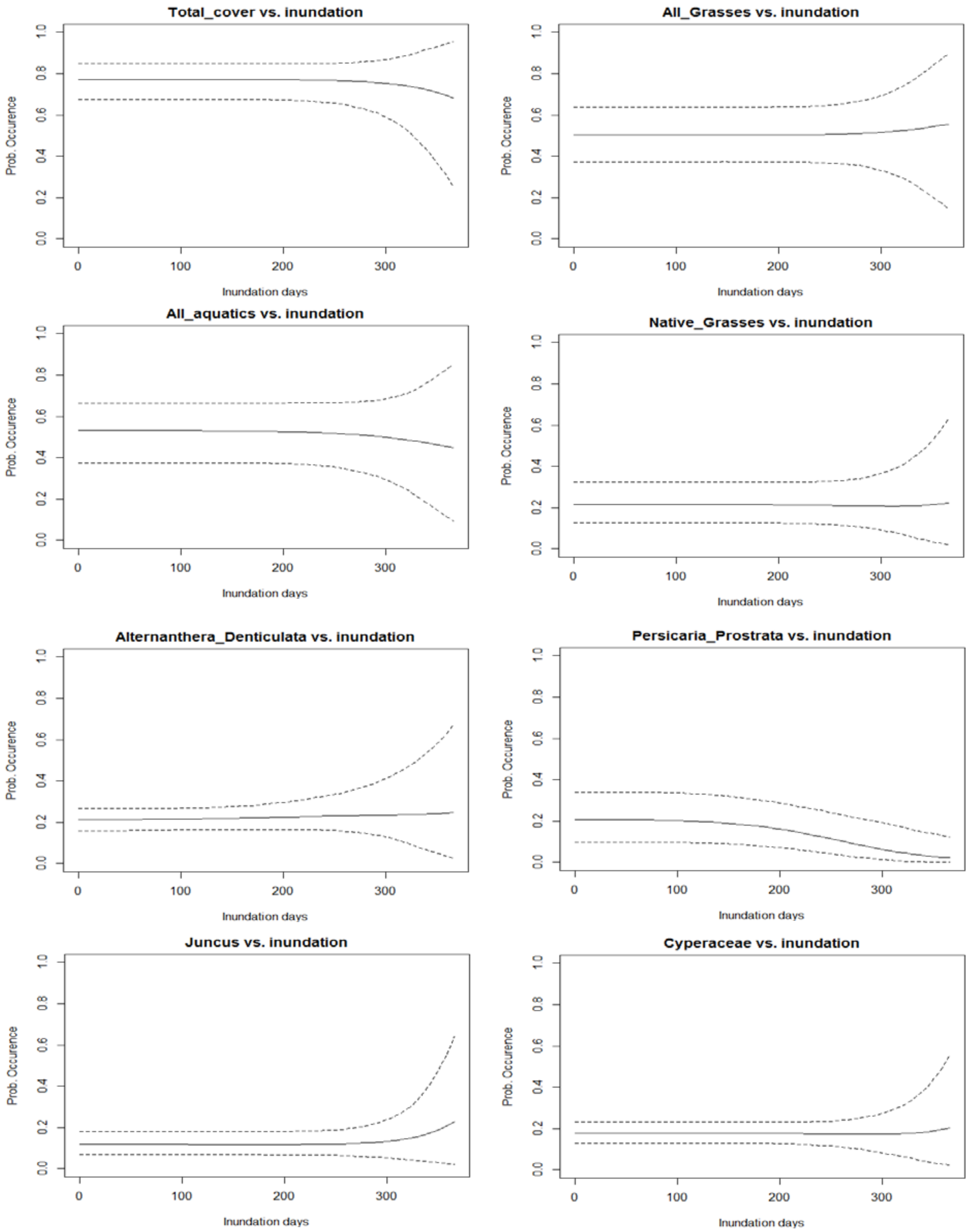


Figure 6-8. Modelled probability of occurrence for different plant groups or species as indicated on graphs grouped in response to number of inundation days in the previous year.

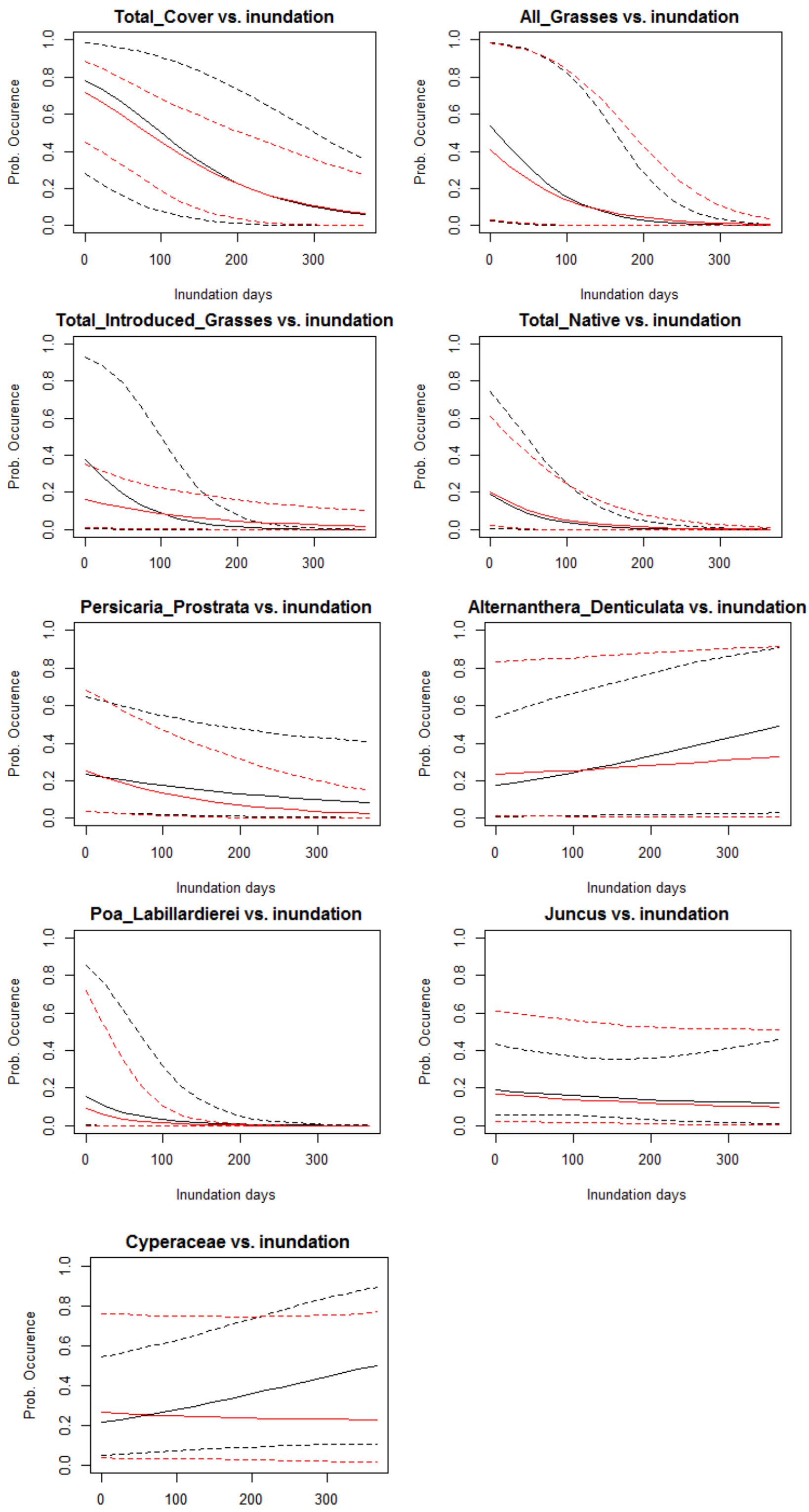


Figure 6-9. Modelled probability of occurrence before fresh (black) and after fresh (red), for grouped water dependent species, in response to number of inundation days in the previous year.

7. Fish

7.1 Introduction

Supporting native fish populations is a key element of the Basin Plan's goal to protect biodiversity. The Goulburn River supports a diverse native fish fauna with high conservation and recreational angling value. Species of conservation significance include trout cod, Murray cod, silver perch, golden perch, Murray River rainbowfish and freshwater catfish. Conservation of the fish fauna of the Goulburn River has been recognised as a high priority by fisheries management and natural resource management agencies. In particular, the provision of environmental flows to support native fish populations has been identified as a key environmental watering objective for the Goulburn River (Cottingham and SKM 2011). Indeed, in terms of Commonwealth water being invested for environmental objectives, flow allocation for native fish represents a major investment of water (e.g. 58 GL for fish habitat maintenance, 138 GL for fish breeding/movement). Given this investment, it is critical that the LTIM Project evaluates the effect that Commonwealth environmental water has on native fish populations in the lower Goulburn River. Quantifying relationships between fish populations (e.g. abundance, distribution, population structure) and environmental flows in the lower Goulburn River will help the adaptive management of environmental flows in the Goulburn River and support decisions regarding environmental flows for fish throughout the Murray-Darling Basin.

The fish monitoring being carried out in this program builds upon 10 years' worth of monitoring and research assessing the status of fish populations in the Goulburn River (Koster et al. 2012) as well as monitoring undertaken since 2006 as part of the Victorian Environmental Flows Monitoring and Assessment Program. When complete, the Goulburn River fish LTIM Project will represent one of the longest continuous sets of fish monitoring data collected in the Murray Darling Basin. Moreover, it will cover a wide range of climatic conditions including record drought, record floods, and a major blackwater event that contributed to widespread fish kills. LTIM project monitoring through to 2018–19 will be particularly important in assessing the ongoing recovery of fish populations from those extreme disturbances.

The Goulburn River fish LTIM Project is also crucial to informing and interpreting the results of monitoring in other parts of the Basin. Golden perch have the capacity to disperse throughout the Basin and there is potentially a high level of connectivity between fish in the lower Goulburn River, lower Murray River, Edward-Wakool system, and Murrumbidgee River (the southern connected Basin). Coordinated monitoring across these four regions is being used to assess the influence of environmental flows in one area (e.g. spawning in the Goulburn River) on fish populations in other areas (e.g. recruitment in lower Murray).

The three fish monitoring methods employed in the Goulburn River LTIM Project (annual adult fish surveys, larval surveys and fish movement) complement each other, and increase the number of evaluation questions and associated research questions that can be answered through the program.

7.1.1 Annual fish surveys

Annual fish surveys in the river channel are part of the LTIM Project Standard Methods for fish monitoring that will provide critical information for the Basin-scale evaluation of Commonwealth environmental water. When added to the existing fish survey data for the lower Goulburn River it will provide a record of how the fish community has changed over a period of 15 years and how those changes relate to river flow. Moreover, annual surveys will help to determine whether fish spawning (detected through larval surveys), or fish movement that may be triggered by environmental flow releases, result in successful recruitment.

7.1.2 Larval fish surveys

The larval surveys for the lower Goulburn River are collecting larvae of all fish species, but will be designed more specifically to detect golden perch spawning. Golden perch is one of only two fish species (along with silver perch) in the Murray Darling Basin for which there is strong evidence of the need for increased discharge to initiate spawning. Indeed, environmental flows in the Goulburn River are explicitly used to promote spawning and recruitment of golden perch; one of the key flow objectives is to deliver freshes to promote the spawning of golden perch (Cottingham and SKM 2011).

The annual adult fish surveys can be used to identify any young-of-year golden perch in the lower Goulburn River, but given golden perch can move long distances, direct egg/larval surveys are required to determine whether high flows released into the lower Goulburn River actually trigger fish spawning.

The larval fish program will build on and add to an existing 10 year data set monitoring the spawning responses of fish to flows in the Goulburn River (Koster et al. 2012). Relatively few golden perch spawning events have been recorded in the lower Goulburn River to date, although we have now seen spawning in three of the four years of the LTIM Project. That is mainly thought to be due to the lack of large flows during the Millennium Drought (2001–2009). The managed flow releases in spring 2013 and 2014 (which used Commonwealth environmental water) triggered the most significant golden perch spawning that has been recorded in the lower Goulburn River in recent years. Ongoing monitoring as part of the LTIM Project should aim to increase knowledge on specific links between key attributes of the flow regime and spawning. This information is critical to help the Goulburn Broken Catchment Management Authority continually refine environmental flows in the future.

The larval fish program will also inform and complement monitoring in other Selected Areas. Fish have the capacity to disperse throughout the Basin and there is potentially a high level of connectivity between regions, particularly the Goulburn, mid- and lower- Murray, Edward-Wakool and Murrumbidgee rivers. That connection means that environmental flows in one area (e.g. spawning in the Goulburn River) has the potential to influence outcomes in other areas (e.g. recruitment in lower Murray). In other words, monitoring of fish spawning responses in the Goulburn River may help to explain changes in recruitment and abundance in other selected areas (and vice versa). Thus, the Goulburn River larval fish LTIM Project will contribute to a comparison and contrast of spawning and recruitment responses of golden perch at sites across much of the Murray Darling Basin, thereby informing Basin-level responses.

7.1.3 Fish movement

Biotic dispersal or movement is critical to supporting connectivity of native fish populations, which is a key element of the Basin Plan’s goal to protect ecosystem function. In particular, movement within and between water-dependent ecosystems (i.e. connectivity) can be crucial for sustaining populations by enabling fish to recolonise or avoid unfavourable conditions. For some fish species, movement also occurs for the purposes of reproduction and therefore contributes to the Basin Plan’s goal to protect Biodiversity.

The Goulburn River fish movement program targets golden perch and will build on the existing six-year acoustic telemetry project monitoring movement of native fish in the Goulburn and Murray rivers that was funded by Commonwealth Environmental Water Office (as part of their Short Term Intervention Monitoring Program) and Goulburn Broken Catchment Management Authority (Koster et al. 2012). The Goulburn River fish movement program complements monitoring of fish movement being undertaken as part of the LTIM Project in the Edward-Wakool and Gwydir rivers. In particular, it will enable a comparison and contrast of the movements of native fish at sites across much of the Murray Darling Basin thereby informing Basin-level responses. Fish have the capacity to disperse throughout the Basin and there is potentially a high level of connectivity between regions, particularly the Goulburn, lower Murray, Edward-Wakool and Murrumbidgee rivers. Therefore, the influence of environmental flows in one area has the potential to strongly influence outcomes in other areas. In other words, monitoring of fish movement within the Goulburn River might help to explain changes in fish abundance within other selected areas.

The LTIM Project is providing a unique opportunity to co-ordinate fish movement monitoring across the southern connected Murray-Darling Basin. A focus is to investigate whether individual golden perch move between any of the selected areas over the course of the LTIM project, and considering whether particular flow events triggered or facilitated that movement.

7.2 Area specific evaluation questions

To determine the contribution of Commonwealth environmental water in selected areas, and to improve understanding of the relationship between specific watering actions and ecological objectives for assets, the following questions are being addressed. This information also forms the basis of Basin-scale evaluation – where area-level results are scaled up to the Basin level.

Question	Were appropriate flows provided?	Effect of environmental flows	What information was the evaluation based on?
What did CEW contribute to the recruitment of young-of-year golden perch in the lower Goulburn River?	Spawning coincided with increases in flow in 2017, but no young-of-year fish were subsequently collected.	Golden perch spawned during a within-channel flow pulse in late November 2017 and in early December 2017 during a larger natural flow event. However, no young-of-year fish were collected in the autumn surveys. It is possible that eggs drift downstream into the Murray River. Under this scenario, recruitment may be reliant on immigration of fish from the Murray River into the Goulburn River.	Qualitative observations based on electrofishing and drift netting data
What did CEW contribute to golden perch spawning and in particular what magnitude, timing and duration of flow is required to trigger spawning?	Spawning coincided with increases in flow, including a spring fresh flow release in November 2017. Extended periods of low stable flows in the pre-spawning period may have a negative impact on golden perch spawning.	Golden perch spawned during a within-channel flow pulse in late November 2017 and in early December 2017 during a larger natural flow event. The majority of golden perch eggs were collected during the larger flow event in early December 2017. Silver perch also spawned during the within-channel flow pulse in late November 2017.	Qualitative observations based on drift netting data. Statistical models predicting the likelihood of spawning, and incorporating data from four years of LTIM monitoring plus earlier data have also been developed.
What did CEW contribute to the movement of golden perch in the lower Goulburn River and where did those fish move to?	Movements coincided with increases in flow, including a spring fresh flow release in November 2017.	Golden perch undertook long-distance movements in the lower Goulburn River in 2017, including moving between the Goulburn and Murray rivers. Most long-distance downstream movements coincided with increases in flow, including a spring fresh environmental flow release in November 2017.	Qualitative observations based on telemetry data. Statistical models predicting the likelihood of movement, and incorporating data from four years of monitoring have also been developed.

7.3 Main findings from the fish monitoring program

7.3.1 Findings from 2017–18

The main findings from 2017–18 monitoring can be summarised as:

Annual surveys (electrofishing and netting)

- Three species of conservation significance were collected in the autumn 2018 surveys: Murray cod, silver perch and Murray River rainbowfish.
- Australian smelt was the most abundant species collected in 2018, similar to the results of previous surveys.
- The abundance of most species (e.g. Murray cod, silver perch, Murray River rainbowfish, Australian smelt, carp gudgeon, carp and eastern gambusia) was lower in 2018 compared to 2017.
- Young-of-year (YOY) Murray cod were collected in the 2018 surveys. However, no YOY golden perch or silver perch were collected.
- The introduced Redfin perch was collected in 2018. This represents the first record of this species in the four years of LTIM Project surveys, although it is not a new record for the system.

Surveys of eggs and larvae (drift nets)

- Golden perch eggs were collected in late November 2017 coinciding with a spring fresh environmental flow release and in early December 2017 during a larger natural flow event.

- The majority (67%) of golden perch eggs were collected during the larger flow event in early December 2017.
- Silver perch eggs were also collected during the spring fresh environmental flow release in late November 2017.
- Trout cod larvae were collected for the first time in the four years of surveys
- Low numbers of carp larvae (n = 16) were collected coinciding with the natural flow event in early December 2017.

Movement of golden perch

- Nine golden perch undertook long-distance movements (i.e. > 20 km) in the lower Goulburn River in 2017.
- Movement was most prevalent between September and December 2017, and occurred primarily in a downstream direction, typically followed by return upstream movements.
- Eight of these fish moved downstream into the Murray River in 2017. Of these fish, seven returned to the Goulburn River.
- Most long-distance downstream movements coincided with increases in flow, including a spring fresh environmental flow release in early October (primarily for vegetation) and late November (primarily for golden perch spawning) 2017.
- In the 2017 spawning season, the occurrence of golden perch eggs in the drift samples coincided with the movements of tagged fish.

7.3.2 How these build on findings from years 1 to 3

These findings build on findings from years 1 to 3 by demonstrating:

- The lower Goulburn River supports several species of conservation significance, including the nationally threatened silver perch and trout cod.
- Murray cod abundance decreased in 2017 following a hypoxic blackwater event in January 2017 (particularly at Zeerust), and declined further in 2018.
- The decline observed in 2018 may have resulted from reduced sampling efficiency associated with elevated flows due to inter-valley-transfers throughout autumn 2018. Future analyses should incorporate approaches to lessen potential error that might result from changes in capture probabilities (Lyon et al. 2014).
- Silver perch abundance increased considerably in 2017, likely due to immigration of fish from the Murray River, but declined in 2018. This result could indicate that fish migrated back to the Murray River, but could also be related to reduced sampling efficiency in 2018.
- Higher flows in spring-early summer, including the delivery of freshes, can promote spawning of golden perch. Golden perch eggs/larvae have been collected in each year of the LTIM Proejct except 2015. However, abundance hae varied among years. Differences in flow conditions among the years in the pre-spawning period may influence levels of spawning (e.g. extended periods of low stable flows throughout spring may have a negative impact on golden perch spawning).
- Higher flows in spring-early summer, including the delivery of freshes, can promote movement of golden perch. Movement occurred primarily during the spawning season and often coincided with the

presence of eggs in drift samples. The coincident timing of movement and spawning suggests that at least some of the movements are related to reproduction.

- Higher flows in spring-early summer, including the delivery of freshes, can promote spawning of silver perch. Silver perch eggs were collected in each year, except 2015.
- Although golden perch and silver perch spawned in each year except 2015, few or no young-of-year fish have been collected in the autumn surveys. Golden perch and silver perch eggs are semi-buoyant and drift downstream, potentially over large distances. It is possible that eggs drift downstream into the Murray River. Under this scenario, recruitment may be reliant on immigration of fish from the Murray River into the Goulburn River.
- The collection of trout cod larvae in 2017 confirms that breeding populations of trout cod still exist within the lower Goulburn River.
- Higher flows in spring-early summer, including the delivery of freshes, can promote spawning of carp. Carp spawning has been detected in the last three years (2015, 2016 and 2017).

7.4 Methods

7.4.1 Field methods

Annual fish surveys

A detailed description of the sampling methods can be found in the Standard Operating Procedures available as part of the Monitoring and Evaluation Plan (Webb et al. 2018). Briefly, electrofishing was conducted at 10 sites in the Goulburn River during April and May 2018. Sampling was conducted at each site during daylight hours using a Smith–Root model 5 GPP boat–mounted electrofishing unit. At each site the total time during which electrical current was applied to the water was 2880 seconds. Ten fyke nets were also set at each site. Nets were set in late afternoon and retrieved the following morning.

Larval fish surveys

Drift nets were used to collect fish eggs and larvae in the Goulburn River at four sites (Pyke Road, Loch Garry, McCoy's Bridge, Yambuna) every week from October to December 2017 using 3 nets set at each site. The nets were set in late afternoon and retrieved the following morning.

Fish movement

A total of 88 adult golden perch were collected from the Goulburn River and tagged with acoustic transmitters over the period autumn 2014–16. Twenty-one acoustic listening stations have also been deployed in the Goulburn River between Goulburn Weir and the Murray River junction as part of this and other monitoring programs. Four listening stations were also deployed in the Murray River near the Goulburn River junction.

7.4.2 Statistical analysis

Larval fish surveys

The probability of spawning of golden perch was modelled with a hierarchical logistic regression:

$$y_i \sim Bernoulli(probability_i) \qquad \text{Equation 1}$$

$$logit(probability_i) = int + Inc\_temp_i \times Inc\_Q2wk_i \times \ eff.Q_j \times Q_i + eff.site_j + eff.net_k + eff.survey_m \qquad \text{Equation 2}$$

$$Inc\_temp_i = \begin{cases} 1, when\ temp \geq temp_{threshold} \\ 0, when\ temp < temp_{threshold} \end{cases} \qquad \text{Equation 3}$$

$$Inc\_Q2wk_i = \begin{cases} 1, when\ Q2wk \geq Q2wk_{threshold} \\ 0, when\ Q2wk < Q2wk_{threshold} \end{cases}$$ Equation 4

The occurrence of spawning (*y*) for drift net *j* at site *k* during year (or survey) *m* and deployment *i* is driven by a global average across all sites (*int*), plus the effect of discharge (*eff.Q*). However, this effect of discharge is only relevant when temperatures exceed certain levels, as determined by an inclusion term (*Inc_temp*). This is achieved by having the inclusion term equal to 0, unless temperature exceeds a threshold ($temp_{threshold}$), which shifts the inclusion term to 1 (Equation 3). The temperature threshold is fitted within the model. Similarly, another threshold for discharge effect is introduced according to antecedent flow (*Inc_Q2wk*). This inclusion term equals to 0, unless when *Q2wk*, the average daily discharge from three weeks to one week prior to each sampling event, exceeds a threshold ($Q2wk_{threshold}$), which shifts the inclusion term to 1 (Equation 4).

There is a random effect of site (*eff.site*) that acknowledges that local conditions may enhance or retard spawning overall, a random effect of each drift net location (eff.net) to account for the repeated measures taken for each net location, and a random effect of each year (*eff.survey*) to account repeated measures taken in each year.

We have explored the following discharge indicators (Q_j in Equation 2) as model predictors:

- Discharge (ML/day)
- Velocity (reach-average velocity, m/s)

Note that pre-LTIM larvae data from 2010 to 2014 were also included in this analysis (from 2010, when flow data became available).

Fish movement

The fish movement data (2014–2018 data combined) were also analysed with a hierarchical logistic regression (probability of occurrence of downstream movement). The occurrence of movement (both upstream and downstream) was defined as the detection of an individual fish at multiple acoustic listening stations, as repeated detections of a fish at a single listening station does not necessarily imply movement away from a home range. The model structure is as follows:

$$move_i \sim Bernoulli(probability.move_i)$$ Equation 4

$$logit(probability.move_i) = int + eff.Q \times Q_i + eff.day1 \times day_i^2 + eff.day2 \times day_i + eff.temp \times temp_i + eff.Fish_j + eff.Year_k$$ Equation 5

The occurrence of movement (*move*) for fish *j* on day *i* is driven by the global average across all sites in the absence of flow (*int*), the effect of discharge (*eff.Q*), the effect of temperature (*eff.Temp*), and the effect of the time of year (*eff.day1* and *eff.day2*). There is also a random effect of the fish *j* (*eff.Fish*), and a random effect of year *k* (*eff.Year*). This is to take into account the fact that the probability of fish movement can vary depending on the specific hydrological conditions of the year.

eff.Fish was modelled hierarchically, being drawn from a normal distribution with the hyperprior (*mu.eff.fish)* modelled as a function of the fish length (in mm). This is to take into account the fact that young fish tend to move less than mature fish.

$$eff.Fish_j \sim N(mu.eff.fish_j, t.eff.fish)$$ Equation 6

$$mu.eff.fish_j = int.fish + eff.Size \times Size_j$$ Equation 7

We have explored the following discharge indicators (Q_i in Equation 5) as model predictors:

- Discharge (ML/day)
- Velocity (reach-average velocity, m/s)

All data were averaged over a moving 5-day timestep.

Six fish were reported (tag numbers: 59600, 59619, 59621, 59626, 55094, 55111) were last detected in the Murray river and not again in the Goulburn. These fish are not expected to be affected by Goulburn flow after migrating into the Murray river. Records of these fish past the date their migration into the Murray river should be excluded from the dataset. However, due to the time constraints for the current reporting cycle, this will be implemented next year.

All fish tags within the current listening station records were within their expected battery life. However, this should be noted in the analyses for next year due to possible expiration of the tag batteries for fish released in earlier years.

7.5 Results

7.5.1 Annual surveys (electrofishing and netting)

Over 600 individuals representing seven native and four exotic species were collected from the ten electrofishing sites in the Goulburn River in 2018 (Table 7-1). Species of conservation significance collected were Murray cod, silver perch and Murray River rainbowfish. Australian smelt was the most abundant species collected, comprising 49% of the total abundance for all species. The introduced carp was the second most abundant species in 2018, and comprised 21% of the total abundance. The abundance of most species in the samples (e.g. Murray cod, silver perch, Murray River rainbowfish, Australian smelt and carp) was considerably lower in 2018 compared to 2017.

Table 7-1. Numbers of individual fish species collected from the Goulburn River in electrofishing surveys 2015-2018. Asterisk denotes exotic fish species

Species	2015	2016	2017	2018	Total
Silver Perch *Bidyanus bidyanus*	2	5	15	3	25
Goldfish *Carassius auratus**	8	22	14	29	73
Carp *Cyprinus carpio**	107	264	388	145	904
Eastern gambusia *Gambusia holbrooki**	1		5	7	13
Carp gudgeon *Hypseleotris sp*	9	28	18	7	62
Trout cod *Maccullochella macquariensis*	1	4			5
Murray cod *Maccullochella peelii*	79	83	53	36	251
Golden perch *Macquaria ambigua*	29	41	30	30	130
Murray River rainbowfish *Melanotaenia fluviatilis*	128	114	214	88	544
Bony herring *Nematalosa erebi*		3	12	1	16
Redfin perch *Perca fluviatilis**				1	1
Australian smelt *Retropinna semoni*	267	349	538	334	1488
Total number of individuals	**631**	**913**	**1287**	**681**	**3512**

Across the four years of sampling (2015–2018), over 3500 individuals representing eight native and four exotic species were collected from the ten electrofishing sites in the Goulburn River (Table 7-2). Species of conservation significance collected were Murray cod, trout cod, silver perch and Murray River rainbowfish. Australian smelt was the most abundant species collected in all four years, comprising 38-49% of the total abundance for all species across the four years. The introduced carp was the second most abundant species in 2016–2018, and third most abundant species in 2015, and comprised 17-30% of the total abundance across the four years.

A total of 524 individuals comprising three native species and two exotic species were collected from the annual netting surveys in 2018 (Table 7-2). Carp gudgeon was the most abundant species captured comprising 52% of the catch. The introduced eastern gambusia was the second most abundant species captured (24%). Three young-of-year carp were collected. The abundance of most species (e.g. carp gudgeon, Murray River rainbowfish, Australian smelt and eastern gambusia) was considerably lower in 2018 compared to 2017.

Table 7-2. Numbers of individual fish species collected from the Goulburn River in fyke netting surveys 2015–2018. Asterisk denotes exotic fish species

Species	2015	2016	2017	2018	Total
Carp *Cyprinus carpio**				3	3
Eastern gambusia *Gambusia holbrooki**		6	5225	127	5358
Carp gudgeon *Hypseleotris sp.*	170	403	651	272	1496
Golden perch *Macquaria ambigua*	2	3	1		6
Murray River rainbowfish *Melanotaenia fluviatilis*	58	94	152	86	390
Flatheaded gudgeon *Philypnodon grandiceps*		1	2		3
Australian smelt *Retropinna semoni*	9	1	60	36	106
Total number of individuals	**239**	**508**	**6091**	**524**	**7362**

Across the four years of sampling (2015–2018), over 7300 individuals representing five native and two exotic species were collected from the annual netting surveys (Table 7-2). Eastern gambusia Australian smelt was the most abundant species collected, although the majority of eastern gambusia were collected in 2017 from a single site, Stewarts Bridge. High flows in spring 2016 might have facilitated dispersal of eastern gambusia from offstream habitats into the river. The abundance of carp gudgeon and Murray River rainbowfish increased over the first three years, then declined in the fourth year.

Length frequency histograms are presented below for three of the large-bodied species collected: Murray cod, golden perch and silver perch. Data from the first three years is also provided for comparison.

Murray cod

The size of Murray cod collected in the 2018 surveys ranged from 70 mm in length and 4.8 g in weight to 710 mm in length and 5.1 kg in weight (Figure 7-1, Figure 7-2). Nine young-of-year (YOY) Murray cod (i.e. <100 mm in length) were collected. In 2018, fewer fish between 150-400 mm in length were collected.

Figure 7-1. Murray cod collected in the Goulburn River

Across the four years, the size of Murray cod collected ranged from 47 mm in length and 1.2 g in weight to 800 mm in length and 8.9 kg in weight (Figure 7-2). The vast majority of the population were below the minimum legal angling size (550 mm) for Murray cod. Young-of-year (YOY) Murray cod (i.e. <100 mm in length) were collected in each year.

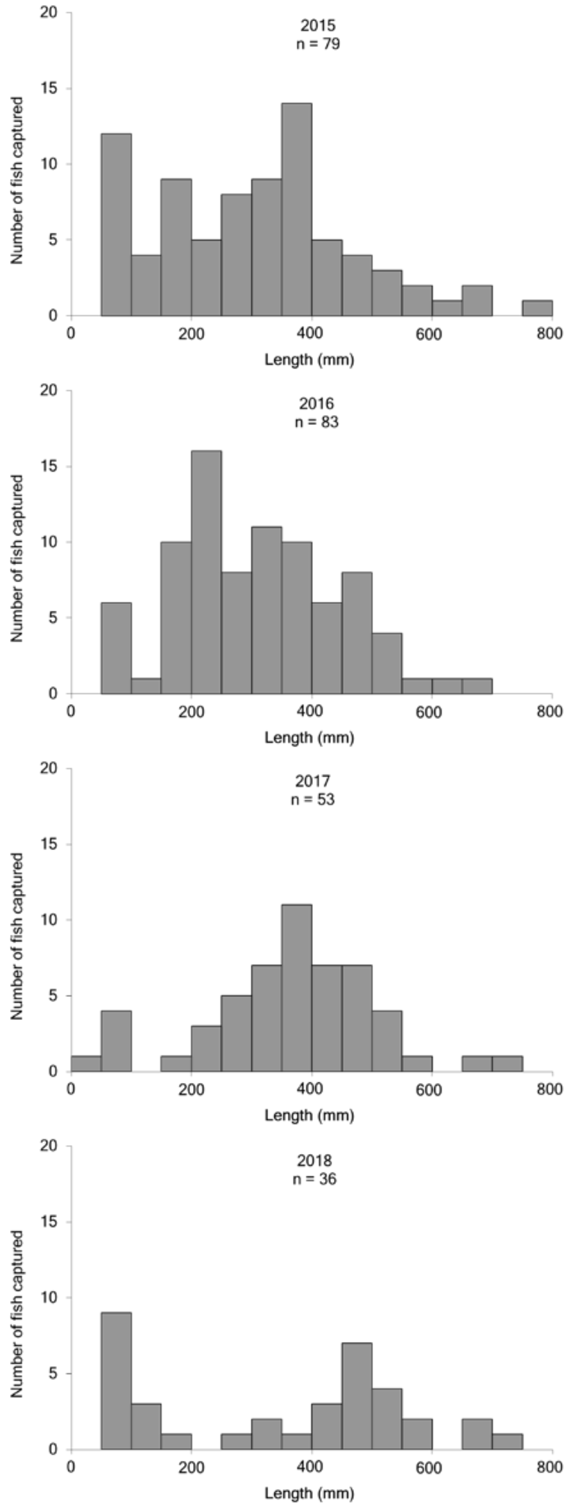


Figure 7-2. Length frequency (total length) of Murray cod collected in the Goulburn River 2015–2018

Golden perch

The size of golden perch collected in the 2018 surveys ranged from 343 mm in length and 530 g in weight to 499 mm in length and 2.3 kg in weight (Figure 7-3). No YOY (i.e. <100 mm in length) golden perch were collected in 2018.

Across the four years, the size of golden perch cod collected ranged from 38 mm in length and 0.6 g in weight to 540 mm in length and 2.6 kg in weight (Figure 7-3, Figure 7-4). The majority of the population consisted of larger, older fish, with few individuals below the minimum legal size of 300 mm. Only three YOY (i.e. <100 mm in length) golden perch were collected, at a single site (Shepparton) in 2016 (Figure 7-3).

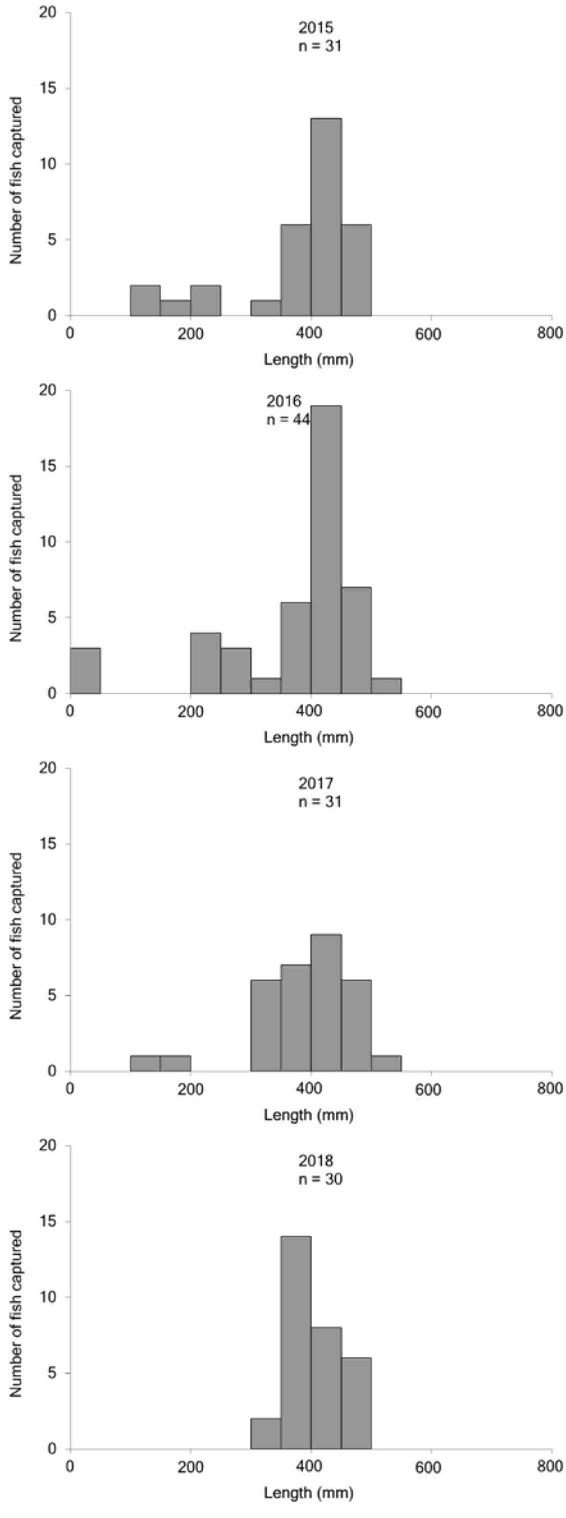


Figure 7-3. Length frequency (total length) of golden perch collected in the Goulburn River 2015–2018

Figure 7-4. Golden perch collected in the Goulburn River

Silver perch

The size of silver perch collected in the 2018 surveys ranged from 274 mm in length and 245 g in weight to 284 mm in length and 3.3 kg in weight (Figure 7-5, Figure 7-6). No YOY silver perch were collected in 2018.

Figure 7-5. Silver perch collected in the Goulburn River

7.5.2 Surveys of eggs and larvae (drift nets)

Over 2600 eggs and larvae representing seven native and two exotic species were collected from the four drift sampling sites in the Goulburn River in 2017 (Table 7-3). Murray cod was the most abundant species collected, comprising 76% of the total abundance for all species.

The drift sampling captured 289 eggs and 11 larvae of golden perch in 2017 (Table 7-3, Table 7-4). Egg and larvae collections coincided with a spring fresh environmental flow release in late November 2017 and a larger natural flow event in early December 2017 (Figure 7-7). The majority (67%) of eggs were collected in early December. Water temperature at these times was 24.5 and 19.5 °C respectively. The drift sampling also captured 118 silver perch eggs (Table 7-3, Table 7-4). Egg collections coincided with a spring fresh environmental flow release in late November 2017 (Figure 7-8).

Across the four years, the size of silver perch collected ranged from 124 mm in length and 20 g in weight to 347 mm in length and 0.6 kg in weight (Figure 7-6). No YOY silver perch were collected, although in 2017 fish between 100-200 mm in length were captured, which are likely 1-2 years old.

Trout cod larvae were collected in early to mid-November 2017 coinciding with a period of low, stable flows. This represents the first time trout cod larvae have been collected in the four years of surveys. Low numbers of carp larvae were collected coinciding with the natural flow event in early December 2017.

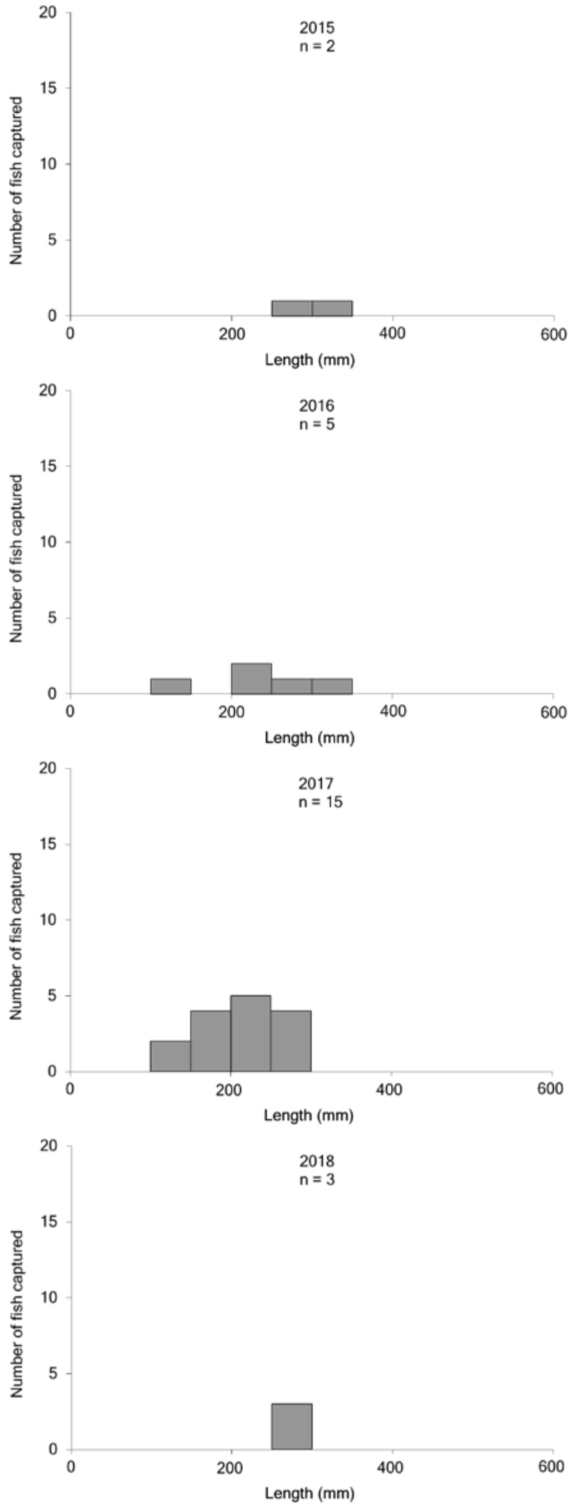


Figure 7-6. Length frequency (fork length) of silver perch collected in the Goulburn River 2015–2018

Across the four years, over 7000 individuals representing seven native and two exotic species were collected from the four drift sampling sites in the Goulburn River (Table 7-3). Murray cod was the most abundant species collected, comprising 60% of the total abundance for all species.

Table 7-3. Numbers of eggs (E) and larvae (L) of fish species collected in drift net surveys from the Goulburn River 2014-2018. Species with asterisk are exotic species.

Species	2014	2015	2016	2017	Total
Silver perch	47E		34E	37E	118
Murray cod	942L	355L	892L	2007L	4196
Trout cod				15L	15
Golden perch	1628E, 1L		47E	289E, 11L	1976
Common carp*		15L	19L	16L	50
Australian smelt	204E, 9L	81E, 7L	32E, 1L	177E, 16L	527
Flathead gudgeon	8L	11L	18L	48L	85
Carp gudgeon		11L	1L	37L	49
Gudgeon sp.				4L	4
Goldfish*				1L	1
Total number of individuals	**2839**	**480**	**1044**	**2658**	**7021**

The drift sampling captured 1964 eggs and 12 larvae of golden perch between 2014 and 2017 (Table 7-3, Table 7-4). Eggs or larvae were collected in each year, but abundance varied considerably between years and sites. Many more were collected in 2014 than in other years, and 82% were collected from the site furthest downstream (Yambuna). None were collected in 2015. Egg and larvae collections coincided with within-channel flow pulses, including environmental watering events in 2014 and 2017, or shortly after bankfull flows, between late October and early December (Figure 7-7). Water temperature around these times varied between 16-27°C.

The drift sampling also captured 118 eggs of silver perch between 2014 and 2017 (Table 7-3, Table 7-4). Eggs were collected in each year, except 2015. Abundance was similar in 2014, 2016 and 2017. Most (88%) were collected from the site second furthest downstream (McCoys Bridge). Egg collections coincided with within-channel flow pulses, including environmental watering events in 2014 and 2017, or shortly after bankfull flows between November and December (Figure 7-8). Water temperature around these times varied between 20-26°C.

Table 7-4. Total number and density (number per 1000 m^3) of golden perch and silver perch eggs and larvae collected during 2014, 2015, 2016 and 2017 sampling events in the Goulburn River.

		2014				2015				2016				2017			
	Stage	Pr	Lg	Mc	Ya	Pr	Lg	Mc	Ya	Pr	Lg	Mc	Ya	Pr	Lg	Mc	Ya
Golden perch																	
Number	Egg	54	314	490	770					1	24	3	19	8	117	100	64
	Larvae			1												1	10
Density	Egg	8.8	17.3	33.5	56.3					0.1	1.9	0.2	0.9	0.1	4.3	3.6	1.5
	Larvae			0.1												0.0	0.2
Silver perch																	
Number	Egg			47								34			6	23	8
	Larvae														0.2	0.8	0.2
Density	Egg			3.2								1.7					
	Larvae																

Pr – Pyke Road, Lg – Loch Garry, Mc – McCoys, Ya – Yambuna

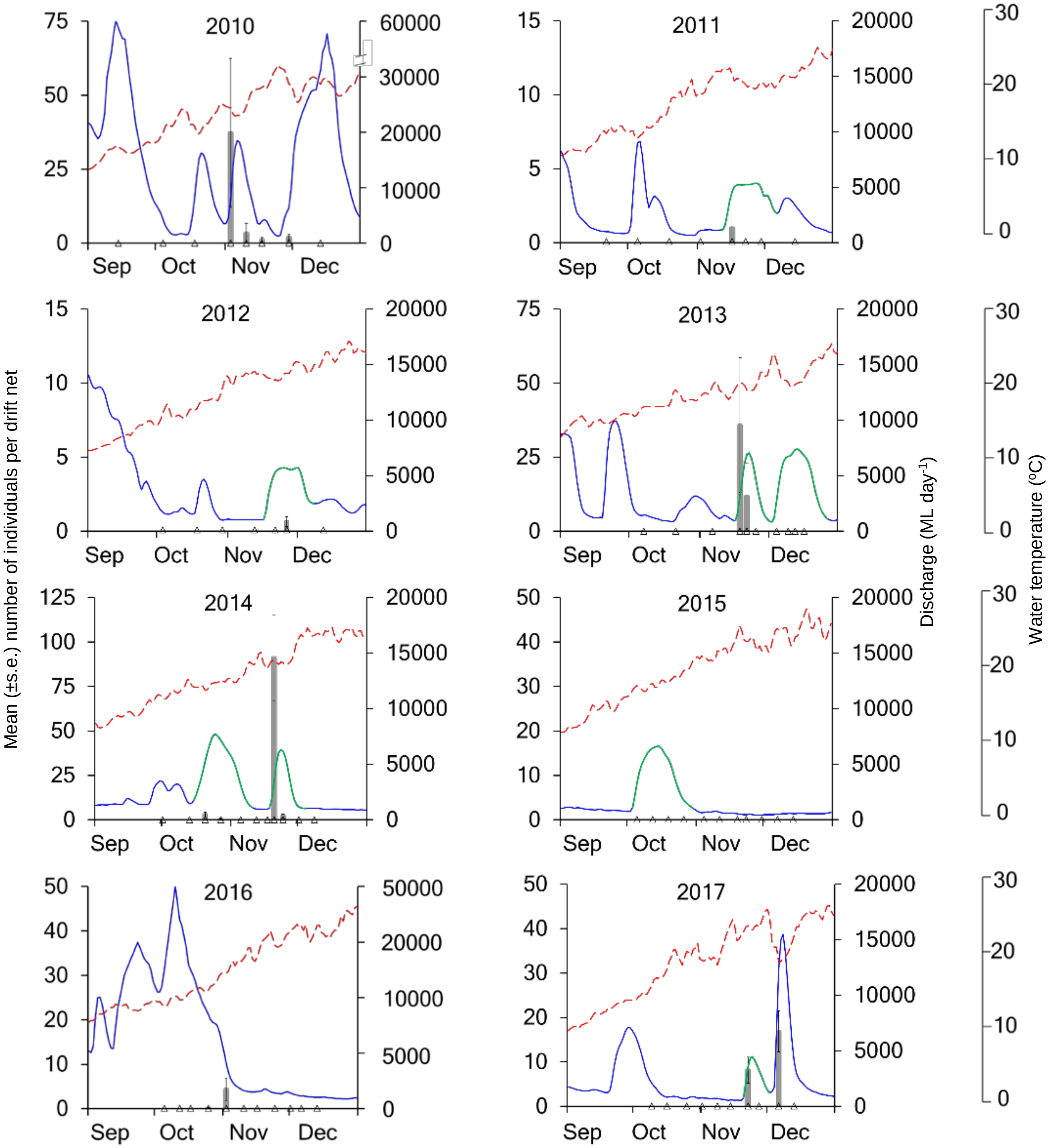


Figure 7-7. Mean (±s.e.) number of golden perch eggs and larvae per drift net collected in the Goulburn River. Mean daily discharge (blue line) and water temperature (broken red line) of the Goulburn River at McCoy's Bridge. Green line denotes environmental flow fresh. White triangles indicate sampling dates. Data from 2010–2013 provided for comparison.

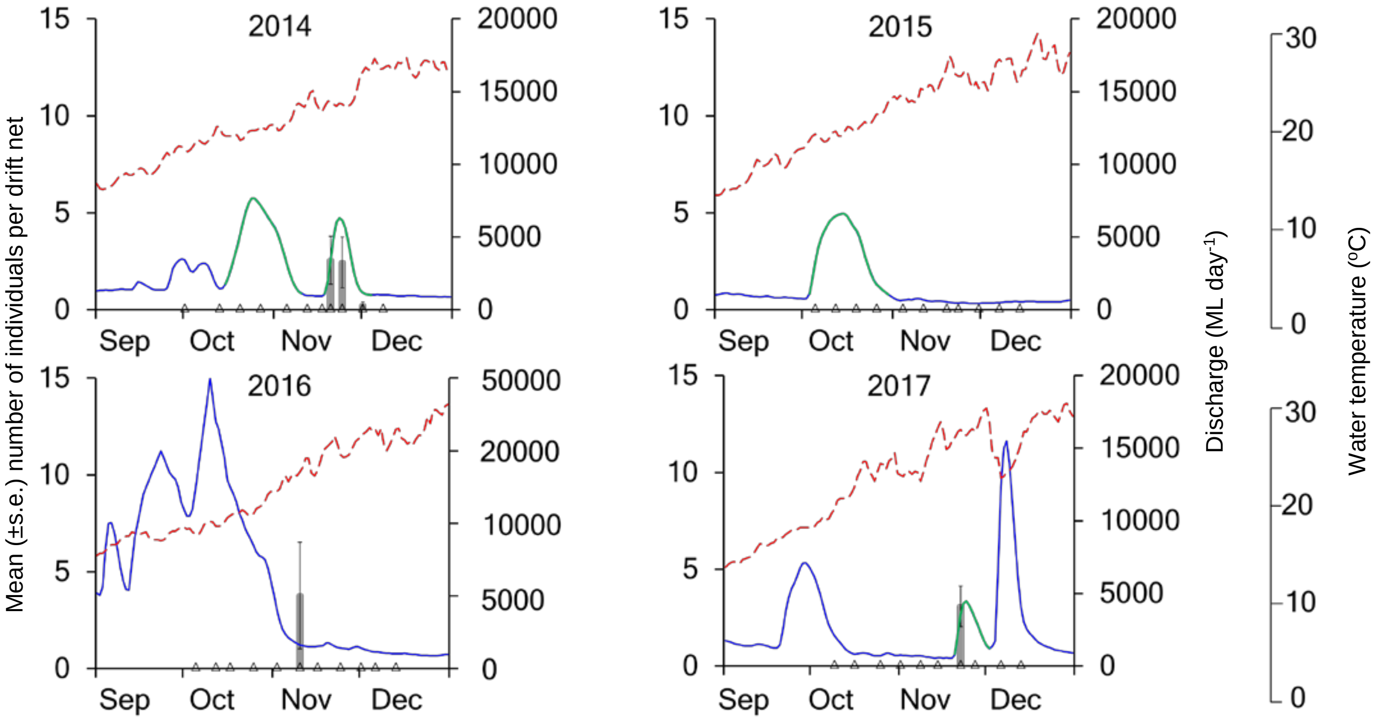


Figure 7-8. Mean (±s.e.) number of silver perch eggs and larvae per drift net collected in the Goulburn River. Mean daily discharge (blue line) and water temperature (broken red line) of the Goulburn River at McCoy's Bridge. Green line denotes environmental flow fresh. White triangles indicate sampling dates.

Statistical analysis

Spawning of golden perch has been modelled as a function of instantaneous flow (Figure 7-9) and velocity (Figure 7-10), with spawning becoming possible when both temperature and antecedent flow exceed certain threshold levels (Section 2.4.1). The fitted temperature threshold is approximately 18.6 degrees for velocity to affect spawning probability. The equivalent, when flow is used as the main predictor, is 23.5 degrees, but with much greater uncertainty (Table 7-5). The calibrated antecedent flow threshold is always smaller than the lowest antecedent flow occurred, suggesting that there is no clear threshold for antecedent flow to affect the impact of flow on spawning probability. The threshold temperature of ~18.5 degrees identified for the velocity analysis is more in line with our previous understanding of spawning in this species. Probabilities of spawning increase with velocity once this temperature has been exceeded.

Table 7-5. Threshold temperature for discharge to impact spawning probability.

Flow			Velocity		
2.5%	Median	97.5	2.5%	Median	97.5
18.51	23.49	23.60	18.37	18.55	18.68

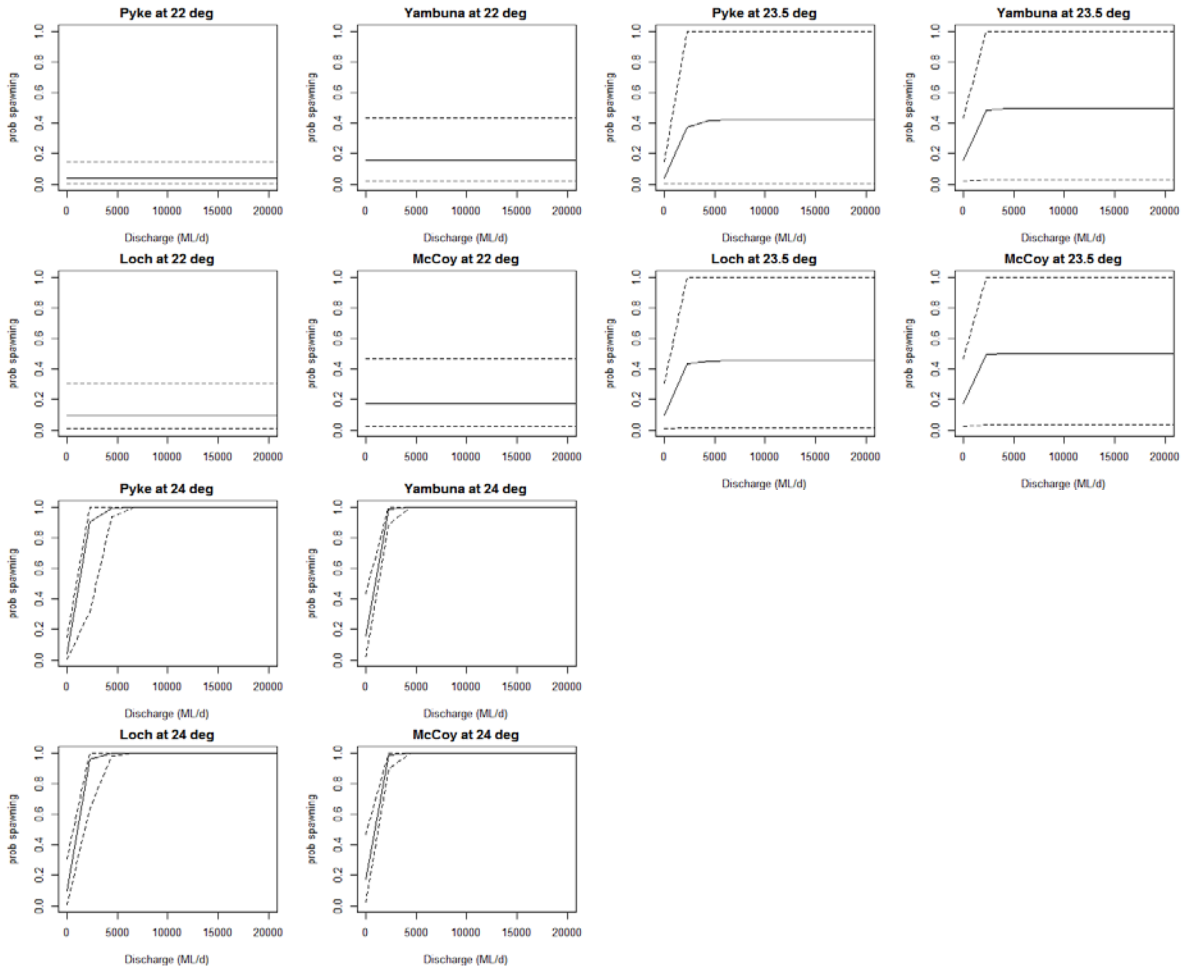


Figure 7-9. Relationship between the probability of occurrence of spawning and discharge (ML/d).

7.5.3 Movement of golden perch

The majority (91%) of the golden perch tagged have been detected by the listening stations. Over half (44 out of 80) of the fish detected have undertaken long-distance movements (i.e. > 20 km); the other 36 fish had no detectable movement (i.e. > 20km) (Figure 7-11, Figure 7-12). Movements by golden perch can generally be grouped into four categories: downstream movement, downstream-upstream return movement, upstream movement, and upstream-downstream movement (Figure 7-11). Some fish displayed multiple modes of movement.

Movement was most prevalent during the spawning season (spring to early summer), and occurred primarily in a downstream direction into the lower river reaches, typically followed by return upstream movements (Figure 7-11, Figure 7-12). Indeed, downstream-upstream return movements were undertaken by almost three-quarters (31 out of 44) of fish, while about one third (15 out of 44) undertook downstream movements (Figure 7-11). In contrast, upstream movement (2 out of 44) and upstream-downstream return movement (5 out of 44) was less common. Twenty-six golden perch (33%) moved downstream into the Murray River. Of these fish, twenty (77%) returned to the Goulburn River.

Most long-distance downstream movements coincided with increases in flow, including spring freshes. In the 2014, 2016 and 2017 spawning seasons, the occurrence of golden perch eggs in the lower reach corresponded with the movements of tagged fish into the lower reaches of the river (Figure 7-12). In 2015, few fish undertook long distance movements, and no golden perch eggs were collected.

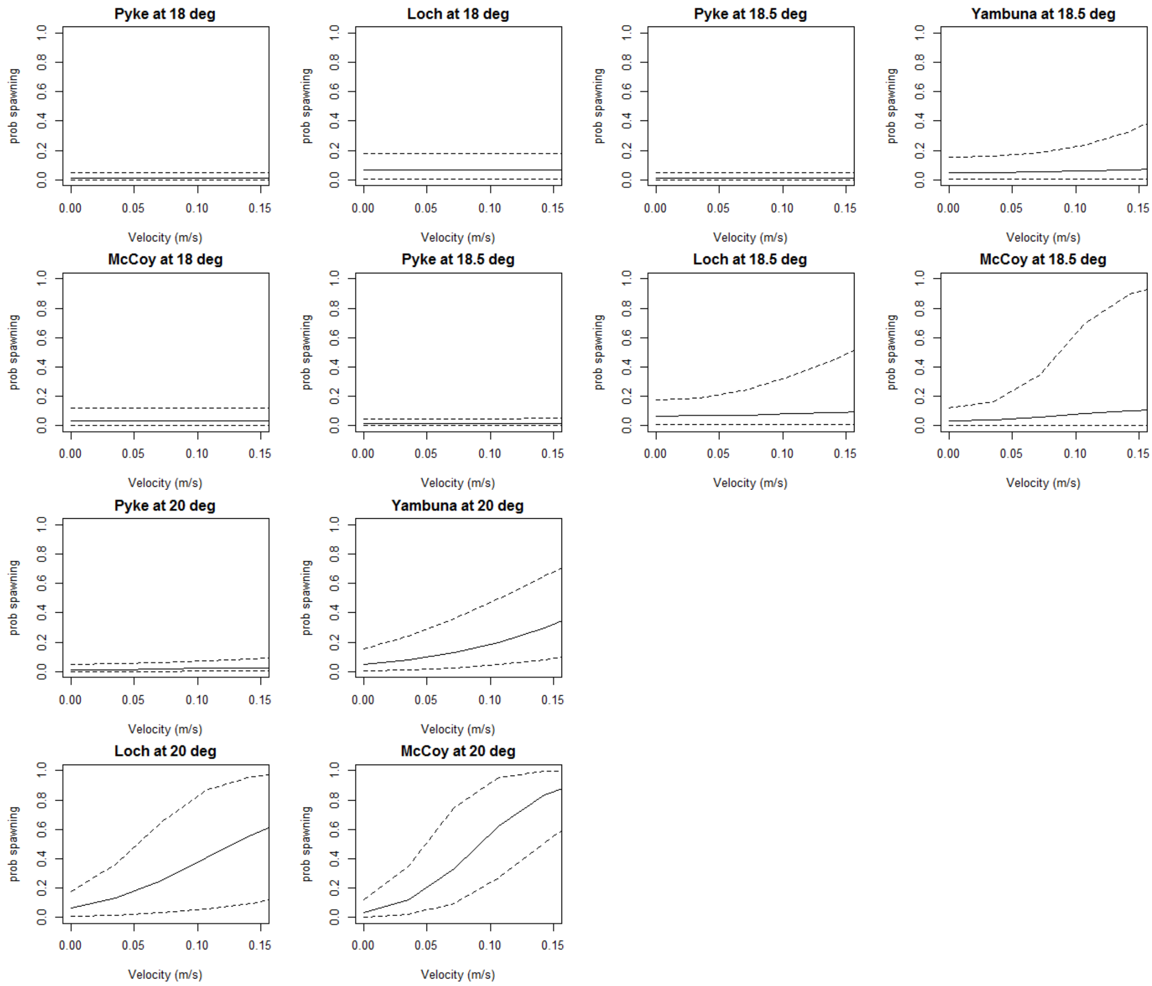


Figure 7-10. Relationship between the probability of occurrence of spawning and velocity (m/s).

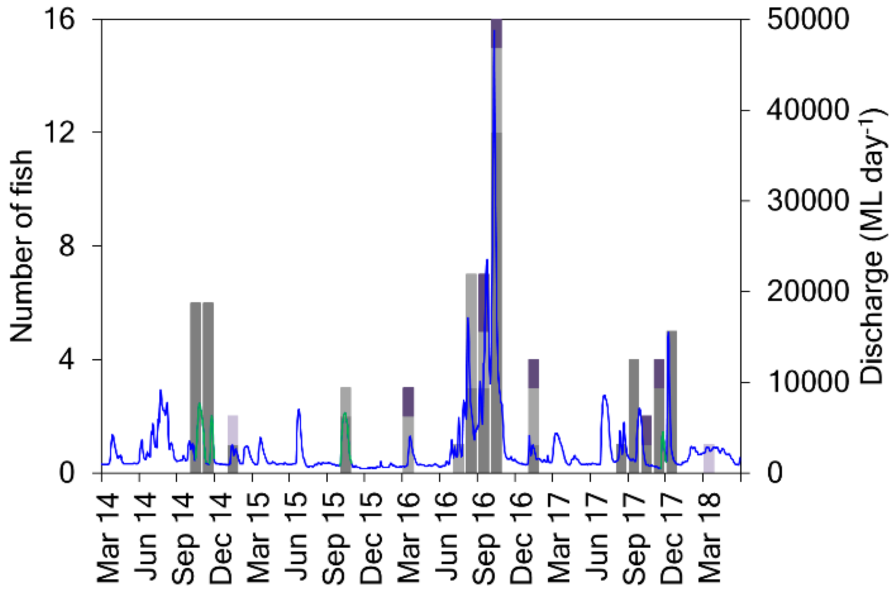


Figure 7-11. Initiation of long-distance movements by golden perch (grouped by month). Coloured bars denote number of individual fish detected moving. Light grey bar = downstream movement, dark grey bar = downstream-upstream return movement, light purple bar = upstream movement, and dark purple bar = upstream-downstream movement. Mean daily discharge (blue line) of the Goulburn River at McCoy's Bridge. Green line denotes spring environmental flow freshes.

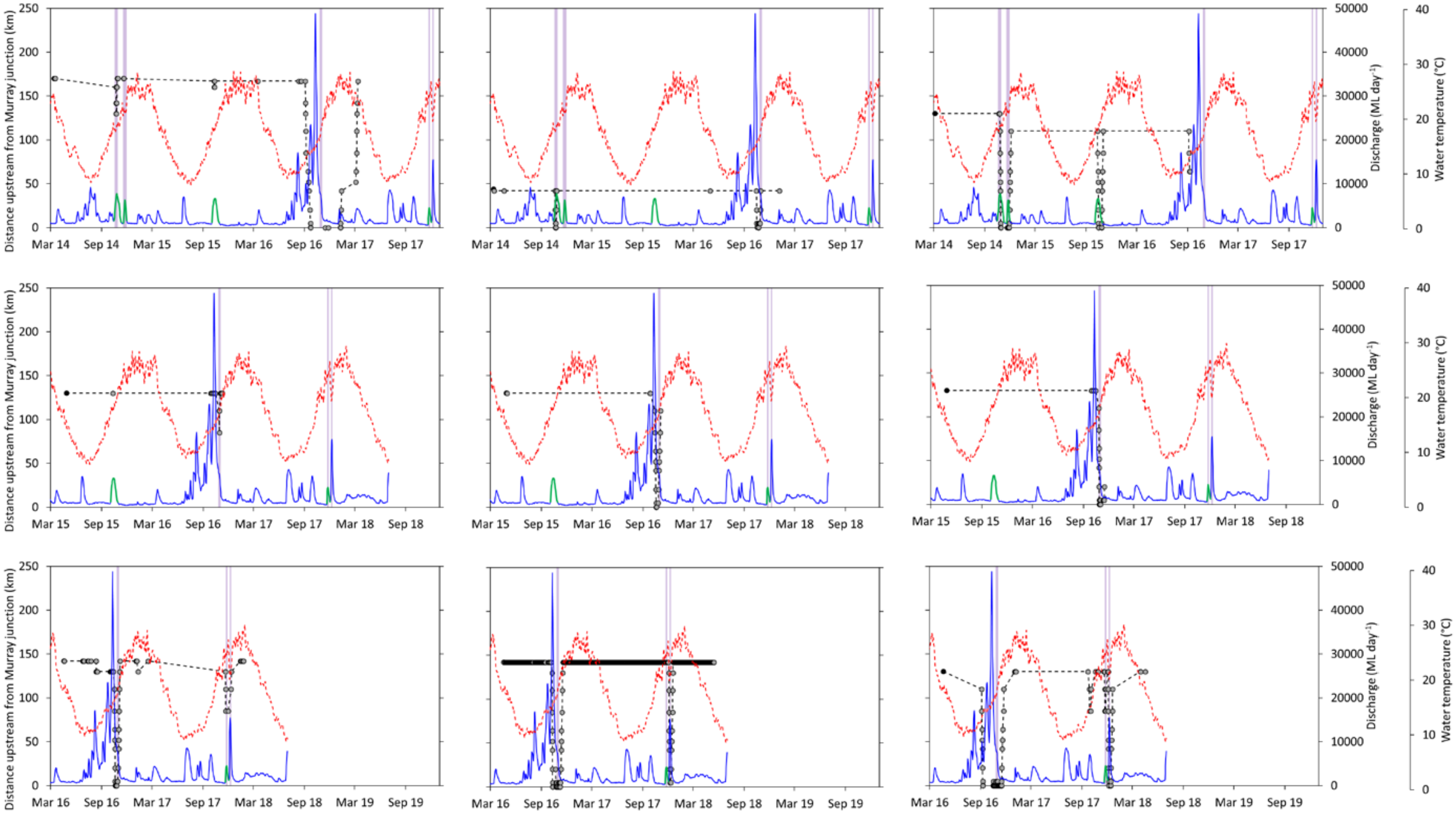


Figure 7-12. Examples of the movement patterns of individual golden perch tagged in the Goulburn River in 2014 (a, b, c), 2015 (d, e, f) and 2016 (g, h, i). Black circles show the date and location of tagging and grey circles show detections of tagged fish on the listening stations. Mean daily discharge (blue line) and water temperature (broken red line) of the Goulburn River at McCoys Bridge. Green line denotes spring environmental flow freshes. Coloured purple bars represent times when golden perch eggs were collected.

Statistical modelling of fish movement indicates that there are significant positive relationships between both paired variables: a) discharge and fish movement; b) temperature and fish movement (Table 7-6). The random effects of any individual years within the four-year period are not significant. The probability that the occurrence of fish movement will increase with environmental flows, due to the increase and change in timing of discharge is 0.58.

Table 7-6. Regression coefficients of fish movement statistical model.

			Flow (ML/d)			Velocity (m/s)
	2.5%	median	97.5%	2.5%	median	97.5%
eff.Q	0.43	0.49	0.54	0.009	0.087	0.165
eff.year1	-9.65	-0.80	8.85	-9.90	-0.04	8.17
eff.year2	-11.76	-2.96	6.63	-12.65	-2.67	5.45
eff.year3	-9.41	-0.56	8.97	-10.75	-0.77	7.37
eff.year4	-9.28	-0.44	9.14	-10.55	-0.55	7.63
eff.temp	0.05	0.20	0.34	0.06	0.20	0.34
eff.Size	-8.70	1.09	9.94	-9.56	0.52	10.89
int	-12.04	-2.03	6.54	-12.39	-2.84	6.66
Prob. increasing movement due to Eflows	0.58	0.58	0.58	0.60	0.60	0.60

As an overall pattern across all years, movement probabilities for individual fish are less than or equal to 0.3 (Figure 7-14). However, movement probability varies depending on the individual fish (Figure 7-13, Figure 7-14). Comparing across individual years, Year 2 shows a clearly lower movement probability (with all probabilities less than 0.1) than all other years.

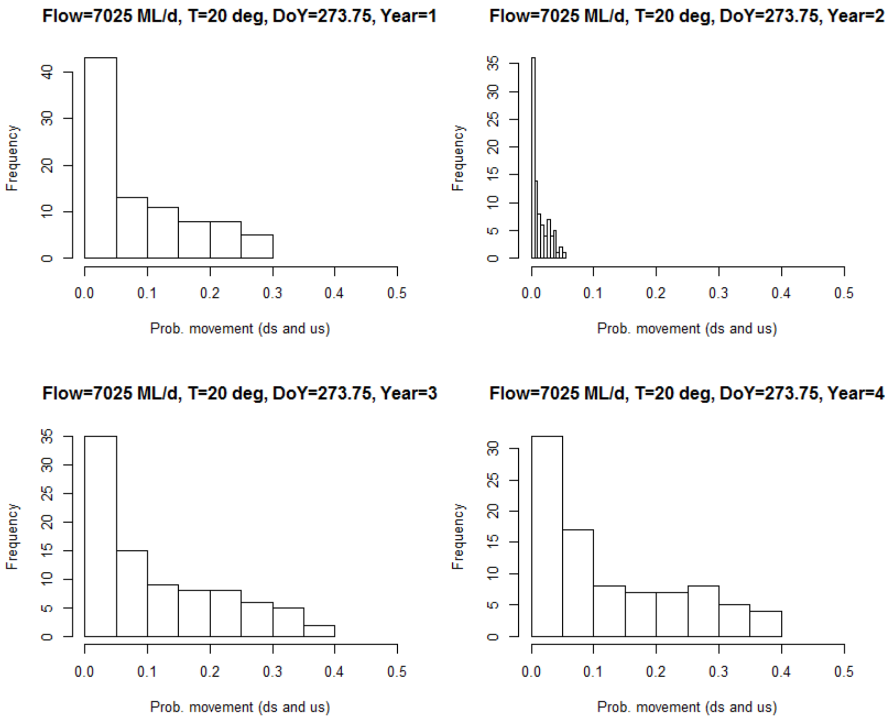


Figure 7-13. Histograms showing the distribution of the average probability of occurrence of movement for each tagged fish, under different flow and temperature conditions, for each of four years.

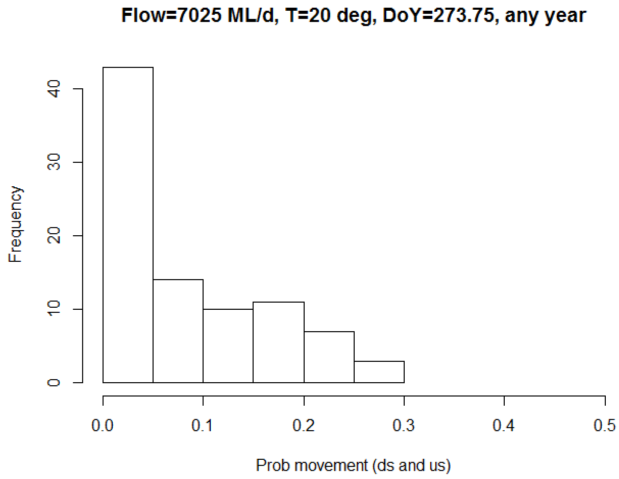


Figure 7-14. Histograms showing the distribution of the average probability of occurrence of movement for each tagged fish, under different flow and temperature conditions, across all four years.

7.6 Discussion

The lower Goulburn River supports several species of conservation significance, including the nationally threatened silver perch, Murray cod and trout cod. An important finding of the recent surveys was the confirmation that breeding populations of trout cod still exist within the lower Goulburn River. Although adult trout cod have not been collected in the annual surveys in the last two years, trout cod larvae were collected at three sites (Pyke Road, McCoys Bridge and Yambuna) in 2017, implying that adults must be present. This collection represents the first record of trout cod larvae in the lower Goulburn River in the four surveys of LTIM Project and in drift surveys since 2008 (Koster et al. 2012). Trout cod are most common in upstream reaches near Murchison (Koster et al. 2012, Tonkin et al. 2017a). The collection of trout cod larvae in 2017 could indicate a recent range expansion, notwithstanding trout cod were not recorded in the annual surveys in the last two years (2017 or 2018). It is also possible that trout cod larvae have drifted downstream from upstream reaches (e.g. Murchison). Targeted monitoring would be valuable to establish the spatial distribution of trout cod in the lower Goulburn River.

Murray cod abundance decreased in 2017 following a hypoxic blackwater event in January 2017 as a result of debris-rich run-off from tributaries (e.g. Castle and Seven Creeks) that entered the Goulburn River near Shepparton. The decline in 2017 was most evident at Zeerust near Shepparton. Decreases in Murray cod abundance were also observed in other LTIM Selected Areas (e.g. Edward-Wakool, Lachlan). Several other fish kills have also occurred in the past in lower Goulburn River, with a large kill occurring in 1984 (Anderson and Morison 1988) and more recent kills occurring in 2004 and 2010 (Koster et al. 2012). It is possible that some Murray cod emigrated from the Goulburn River to avoid the adverse water quality in January 2017 (see for example Leigh and Zampatti 2013, Tonkin et al. 2017b). Murray cod abundance in the lower Goulburn River remained suppressed in 2018, and indeed declined further compared to 2017. The decline in numbers of Murray cod in 2018 was more widespread throughout the Goulburn River. This may have resulted from reduced sampling efficiency associated with elevated flows due to inter-valley-transfers throughout autumn 2018. Supporting this hypothesis, the abundance in samples of numerous other species such as silver perch, Murray River rainbowfish, Australian smelt, carp gudgeon, carp and eastern gambusia was also reduced in 2018. Future data analyses should incorporate approaches to lessen potential error that might result from changes in capture probabilities (Lyon et al. 2014).

Silver perch abundance increased considerably in the Goulburn River in 2017. Silver perch abundance increases were also observed in the nearby Campaspe River in 2017 (Tonkin et al. 2017a). High flows in late 2016–early 2017, including an autumn fresh environmental flow release in 2017, may have promoted immigration of silver perch into these rivers. As part of an MDBA/VEFMAP project, silver perch tagged in the Murray River at Torrumbarry Weir were detected moving upstream and into the Goulburn River in autumn 2017 coinciding with an autumn fresh (Tonkin et al. 2017a). Silver perch abundance in 2018 declined. Silver Perch are a highly mobile species (Mallen-Cooper et al. 1996, Tonkin et al. 2017c). Indeed the tagging study showed that some silver perch returned to the Murray River, usually as flows in the Goulburn River receded (Tonkin et al. 2017a). The reduced abundance of silver perch in 2018 could also be related to reduced sampling efficiency inferred in 2018. Although the increase in silver perch abundance may have been temporary, fish that migrate nonetheless may have had access to increased food availability, which can lead to greater growth and increased fitness (Gillanders et al. 2015, Tonkin et al. 2017c).

Higher flows in late October-early December, including bankfull flows and within-channel freshes, can promote spawning of golden perch and silver perch. Golden perch and silver perch spawned in each year except 2015. In the case of golden perch, many more eggs were collected in 2014 than in the other years. Differences in flow conditions among the years in the pre-spawning period may influence levels of golden perch spawning. For example, in years characterised by extended periods (e.g. 2-3 weeks) of low stable flows throughout spring (e.g. 2011, 2012, 2015, 2017), little or no spawning occurred on the first subsequent flow pulse. Food availability is known to influence reproductive development in female golden perch with a decrease in food reducing reproductive development (Collins and Anderson 1999). It is possible therefore that during extended low flow conditions, food may be more limiting, hence a reduction in spawning.

Higher flows in spring-early summer, including bankfull flows and within-channel freshes, can also promote movement of golden perch within the Goulburn River, and between the Goulburn and Murray rivers. Movement occurred primarily downstream during the spawning season and often coincided with the presence of eggs in drift samples. About 33% of the fish moved into the Murray River. These movements were mostly characterised by temporary occupation, with fish returning to the Goulburn River. However, about 23% of these fish did not return to the Goulburn River. The coincident timing of movement and spawning suggests that movements are related to reproduction. Obtaining finer scale information on golden perch movement and spawning behaviours, using techniques such as radio-telemetry which can provide very high spatial resolution (i.e. <2 m), coupled with direct evidence of spawning (e.g. collections of eggs and larvae) would be valuable to identify potential spawning areas and whether there are particular instream characteristics associated with such areas. The findings also highlight the potential for movement to influence population dynamics and demography in other regions and the need for a river-scale perspective for the management of golden perch such as using coordinated environmental flows across broad spatial scales.

Although golden perch and silver perch spawned in each year except 2015, few or no young-of-year fish were collected in the autumn surveys. Golden perch and silver perch eggs are semi-buoyant and drift downstream, potentially over large distances. It is possible that eggs drift downstream into the Murray River. Under this scenario, recruitment may be reliant on immigration of fish from the Murray River into the Goulburn River. Determining whether golden perch and silver perch in the Goulburn River have migrated into the system from elsewhere, and relating this to patterns of flow, is currently being investigated as part of the LTIM, VEFMAP and EWKR projects using otolith chemistry and telemetry. There also remains the possibility that adult populations of golden perch in the Goulburn River are being maintained by stocking. The identification of adults with a Murrumbidgee water signature in their otoliths (W. Koster, unpubl.) provides evidence that stocked fish make up at least part of the adult population, but adult golden perch are being found with otolith signatures from several different rivers.

Higher flows in spring-early summer, including bankfull flows and freshes, can promote spawning of carp. Carp spawning has been detected in the last three years (2015, 2016 and 2017). Recent modelling indicates that within-channel flows may have relatively little effect on carp recruitment, whereas widespread flooding can lead to substantial recruitment (Koehn et al. 2017). Indeed, large carp abundance increases were observed following the 2016 flooding across multiple Selected Areas (Stoffels et al. 2017). Widespread flooding in spring 2016 likely facilitated substantial spawning events at some locations, resulting in recruitment of carp to the reach.

The introduced Redfin perch was collected in 2018, which represents the first record of this species in the four years of surveys. This species was once common and widespread in the lower Goulburn River, but has declined since the 1980s, possibly as a result of an outbreak of epizootic haematopoietic necrosis in the 1980s (CSIRO 2002).

8. Stakeholder communications

The following communication and engagement actives were undertaken over the 2017–18 period to inform stakeholders and the broader community about the aims and results of the Goulburn River LTIM Project and the role of the Commonwealth Environmental Water Office in environmental water management. Selected examples of communications are included below.

8.1 Media Releases and Articles

Between July 2017 and June 2018 12 media releases were prepared and monthly columns/advertisements were run in the *Shepparton Advisor* and the *Country New*s. These promoted the project, Commonwealth environmental water use in the Goulburn River and ecological responses (native fish movement and breeding, bank vegetation growth and bank erosion) to environmental flows. These resulted in 10 corresponding articles published in local newspapers including the *Shepparton Advisor, Euroa Gazette, Alexandra Standard, Riverine Herald* and the *Country New*s. Articles were also included in the GB CMA electronic newsletter *Connecting Community and Catchment*, which has over 900 subscribers.

Angus Webb led an article in *The Conversation* that described outcomes from the whole LTIM Project (https://theconversation.com/it-will-take-decades-but-the-murray-darling-basin-plan-is-delivering-environmental-improvements-93568) as an attempt to counter some of the negative reporting of the Basin Plan emerging in the media. This article has been read approximately 10,000 times. Following the article, Angus was interviewed by Peter Hannam, Environment Editor for the Sydney Morning Herald, but no specific publication came out of that interview.

Angus was also interviewed for an article in The University of Melbourne's Pursuit magazine (https://pursuit.unimelb.edu.au/articles/repairing-the-murray-darling-basin) that was more closely focused on the Goulburn River LTIM Project.

8.2 Technical publications

Over 2017–18, several publications have appeared in the peer-reviewed scientific literature that report on aspects of the Goulburn River LTIM Project and the greater LTIM Project to a greater or lesser extent.

- Horne AC, Webb JA, Stewardson MJ, Richter B, Acreman M (Eds) (2017) *Water for the Environment: from Policy and Science to Implementation and Management*. Elsevier, Cambridge MA.
- Vietz GJ, Lintern A, Webb JA, Straccione D (2018) River bank erosion and the influence of environmental flow management. *Env. Manage.* **61**, 454-468.
- Stewardson MJ, Gaurino F (2018) Basin-scale environmental water delivery in the Murray-Darling, Australia: a hydrological perspective. *Freshw. Biol.* **63**, 969-985

8.3 Social Media

Numerous Facebook posts and tweets promoted the project and the benefits of environmental water. These were viewed thousands of times and are usually amongst GB CMA's most popular and engaging posts. Currently, the GB CMA has over 3,000 social media followers.

- https://www.facebook.com/gbcma
- https://twitter.com/gbcma

8.4 Videos

The short web videos developed to explain environmental water, blackwater and each key monitoring activity (fish, vegetation, macroinvertebrates, stream metabolism and bank condition) continue to be viewed on a regular basis. The videos have collectively been viewed 473 times in the 2017–18 period. Two new web videos were developed in the 2017-18 period. One on the social, economic and recreational benefits of environmental water in the Goulburn and Broken Catchments (June 2018) and one on the benefits of a higher mid Goulburn River winter baseflow provided by environmental water releases (August 2017). These have been viewed 135 and 185 times respectively.

- Shared Benefits: https://www.youtube.com/watch?v=Pk9w4prdNno
- Goulburn Flows: https://www.youtube.com/watch?v=gtCTOnCSx54&feature=youtu.be

8.5 Presentations

GB CMA staff presented/provided updates to a number of government, community and agency groups throughout the year on environmental water management and the Goulburn River LTIM project. These groups included:

- MDBA graduates;
- South Australian MDB Royal Commission;
- Commonwealth House of Representative enquiry into Commonwealth Environmental Water;
- Victorian Parliament enquiry into environmental water;
- Yorta Yorta Nation Aboriginal Corporation;
- Taungurung Aboriginal Corporation;
- Parks Victoria;
- Goulburn-Murray Water;
- Schools and research institutes
- Australia China Sustainable Agricultura Forum;
- Shepparton Ethnic Council;
- Recreational fishing groups and fish management agencies;
- River Basin Management Society;
- GB CMA partnership group;
- Environmental Water Advisory Groups; and
- Fairley Leadership Group.

Angus Webb presented talks on different aspects of the Goulburn River and larger LTIM Project at international conferences

- International Society for River Science, Hamilton, New Zealand, Nov 2017
- Society for Freshwater Science, Detroit, USA, May 2018
- International Society for Ecohydraulics, Tokyo, Japan, August 2018

8.6 Examples of media

Figure 8-1. Shepparton News 13 September 2017

Freshening flows for river

By Taylah Burrows

An environmental flow along the Goulburn River, between Goulburn Weir and Murray River, will start flowing on Saturday.

"Much of the rain and run-off into the Goulburn River is now captured in the dams and used to supply towns, industry and farms, the amount of water flowing down the river in spring has reduced," Goulburn Broken CMA environmental water manager Simon Casanelia said.

"It also means the river flows higher and faster in the hotter months of the year when communities require more water, which is the opposite of what would happen if there were no dams and weirs."

Mr Casanelia said the changes had affected the health and survival of native plants and animals.

. . . THE AMOUNT OF WATER FLOWING DOWN THE RIVER IN SPRING HAS REDUCED.

— SIMON CASANELIA

He said the flows would be a helping hand, delivering water to mimic more natural flow conditions for the time of the year.

Water for the increased flow, peaking at 8500 megalitres a day, will be released from Goulburn Weir on Saturday.

The increased flows will take about four days to reach McCoys bridge, near the Murray River, and will be well below minor flood levels.

The river will gradually rise to about 4 m at Murchison, 5.3 m at Shepparton and 5 m at McCoys bridge, before slowly returning to current levels by mid-October.

Mr Casanelia said the environmental flow would help bank-stabilising plant growth on the lower banks of the lower Goulburn River and improve water quality and provide food and shelter for waterbugs and native fish.

He said irrigators would also appreciate improved water quality, while people out enjoying activities on and by the water will benefit too.

Dam good: Environmental flow will mimic natural flow conditions for this time of year. Photo: contributed

Figure 8-2. Country News 14 November 2017

4 Country News week of Tuesday, November 14, 2017 www.countrynews.com.au

Enviro flow aims to help fish

Up to 37 Gl will be delivered along the lower Goulburn River this month in an environmental flow designed to benefit golden and silver perch populations.

The flow will see Goulburn-Murray Water release about 5 Gl to meet minimum river operation requirements with a maximum of 23 Gl of environmental water needed with the aim of triggering native fish breeding and improving overall river health.

Commonwealth Environmental Water Holder David Papps said it was important to continually use science to understand how plants and animals were affected when environmental water was delivered, describing native fish as a "barometer" of river health.

Monitoring shows that the best spawning response by golden and silver perch was usually later in spring during a higher flow when water temperatures had started to rise.

"Over the years we have refined the timing of flows to get the best spawning response, with mid to late spring being the most effective," Goulburn Broken Catchment Management Authority's environmental water manager Simon Casanelia said.

"The last time we were able to deliver an environmental flow at this time of the year was 2014 — as last year natural flooding was occurring, and the previous year due to the very dry conditions and low water availability the environmental flow was cancelled."

Water for this spring's environmental flow will peak at 5.5 Gl/day and is due to be released from Goulburn Weir from November 15.

The increased flows will take about four days to reach McCoys Bridge near the Murray River with rivers to gradually rise to about 2.9 m at Murchison, 4.1 m at Shepparton and 3.8 m at McCoys Bridge before slowly returning to current levels (0.8 m at Murchison, 2.8 m at Shepparton and 1.5 m at McCoys Bridge) by late November.

These increases in river flow and height will be well below minor flood level.

In the event of heavy rain, the timing and size of the environmental flow could change or not go ahead at all.

The environmental flows will also act as a trigger for other native fish to move, breed and find shelter; provide increased shelter for water bugs; and improve water quality.

"This all helps crayfish, shrimps, water bugs and native fish continue to recover after the naturally occurring blackwater event that happened earlier this year after a summer storm," Mr Casanelia said.

"Irrigators appreciate better water quality too and of course, now the weather has warmed up, more people will be out and about on and by the river boating, fishing, bushwalking and birdwatching."

Based on recent water trades from Waterpool Co-Op which valued a megalitre of water about $100, the cost of the water used in the environmental flow is more than $3 million on the temporary water market in the Goulburn system.

Figure 8-3. Tweet 15 November 2017

Goulburn Broken CMA @GBCMA · 15 Nov 2017

A small #enviroflow that's making its way along the lower #goulburn river aims to trigger silver & golden perch spawning. Last time a flow was delivered at this time of the year (2014) we got a terrific response. More info: bit.ly/2hda4VA

VEWH, Environmental Water, environment.gov.au and 5 others

8 13

Figure 8-4. Tweet 4 March 2018 promoting Angus Webb's article in *Pursuit*

Figure 8-5. Tweet 4 June 2018 promoting fish monitoring results.

Figure 8-6. Facebook post 27 June 2018

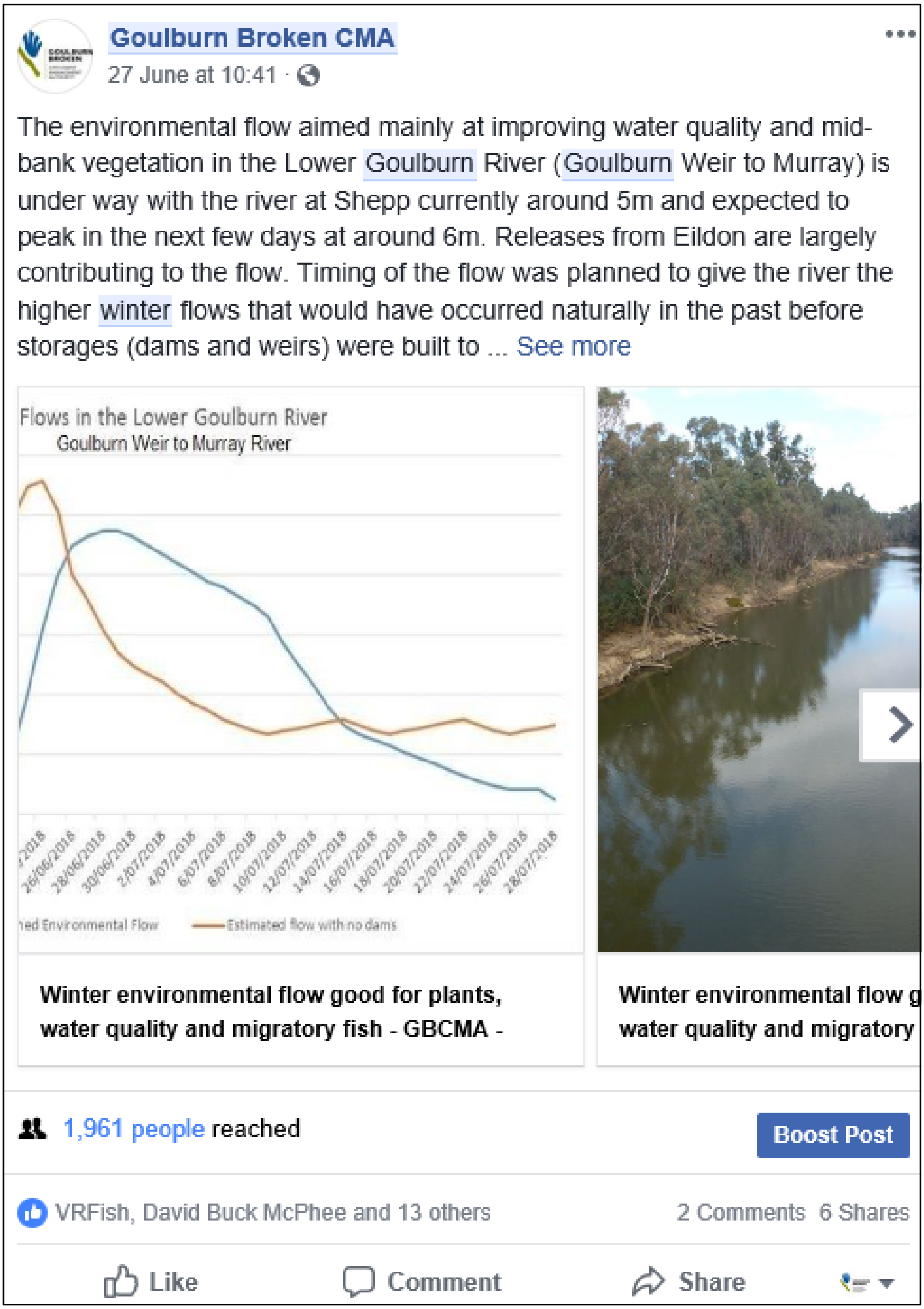


Figure 8-7. Facebook post 21 June 2018

Goulburn Broken CMA
21 June ·

If you spot these astro-turf mats pinned to sections of the lower Goulburn River bank, please don't move them! We're working with scientists and the Commonwealth Environmental Water Office to monitor how seeds, soil and other sediment is deposited and dispersed after flows.
An environmental flow has started this week so once it's over (in about a month) the mats will be collected and analysed (and seeds grown out) to determine what's been deposited and where.
This info will contribute to long-term planning, monitoring and evaluation of the benefits of environmental flows on river health. Find out more about the flow here http://bit.ly/2JQptek
and about monitoring here http://bit.ly/2vV0XOo
@thecewh
Victorian Environmental Water Holder
Goulburn-Murray Water
Greater Shepparton City Council
Parks Victoria
Australian Department of the Environment and Energy
@mdbAuth

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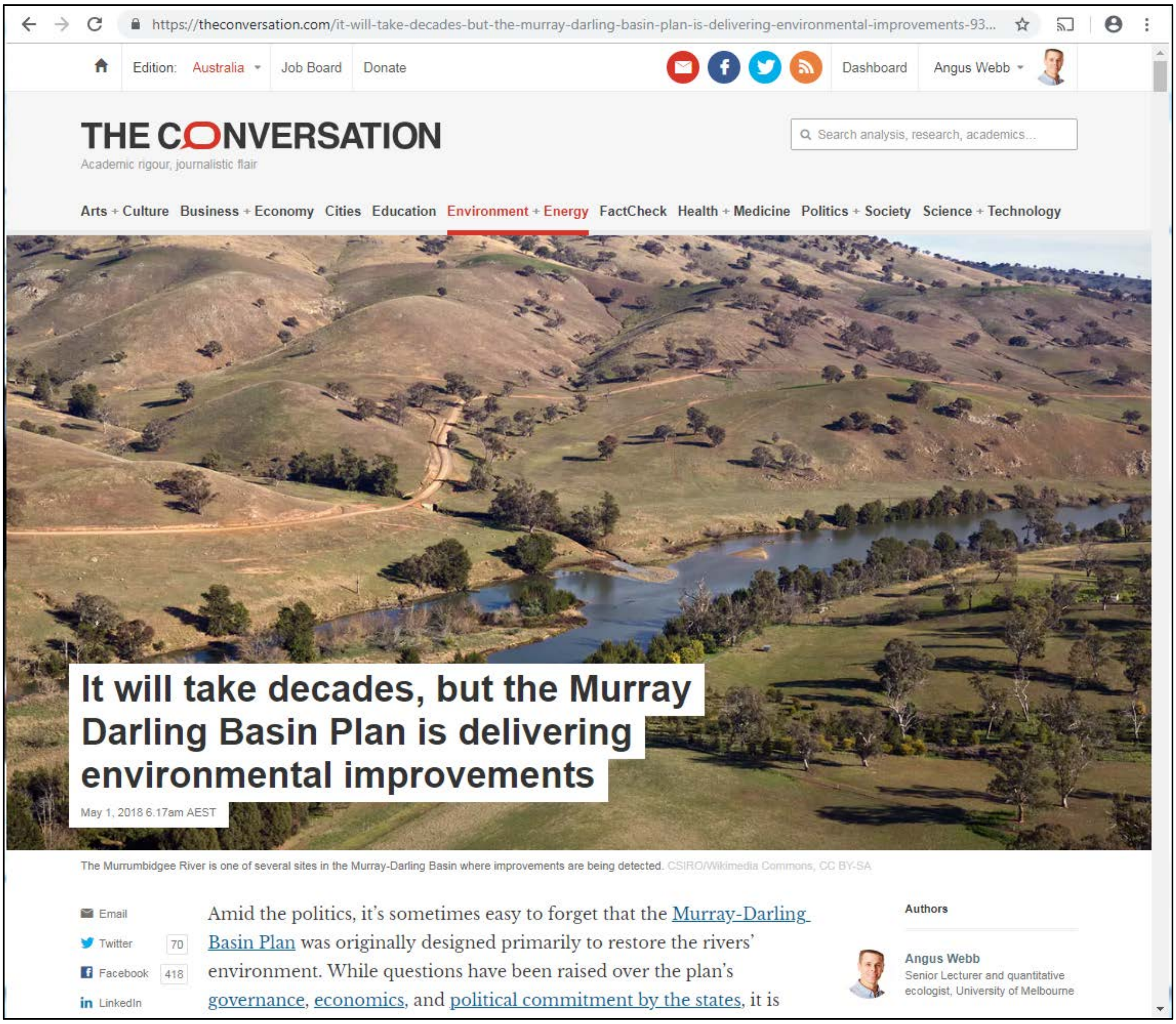
Figure 8-8. Facebook post 16 November 2017

Figure 8-9. Banner shot from the article in *The Conversation*



https://theconversation.com/it-will-take-decades-but-the-murray-darling-basin-plan-is-delivering-environmental-improvements-93...

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It will take decades, but the Murray Darling Basin Plan is delivering environmental improvements

May 1, 2018 6.17am AEST

The Murrumbidgee River is one of several sites in the Murray-Darling Basin where improvements are being detected. CSIRO/Wikimedia Commons, CC BY-SA

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Amid the politics, it's sometimes easy to forget that the Murray-Darling Basin Plan was originally designed primarily to restore the rivers' environment. While questions have been raised over the plan's governance, economics, and political commitment by the states, it is

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