A Report for the Commonwealth Environmental Water Office by The University of Melbourne

Monitoring of ecosystem responses to the delivery of environmental water in the lower Goulburn River and Broken Creek in 2012-13

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EXECUTIVE SUMMARY

Goulburn River

Commonwealth environmental watering

A total of 201,097 megalitres of Commonwealth environmental water was delivered in the Goulburn River during the 2012-13 water year, primarily targeting the maintenance of vegetation and fish condition. Commonwealth environmental water contributed to:

- two freshes in mid October and mid November 2012 of approximately 5,000 - 6,000 megalitres per day at peak magnitude
- two freshes in early and late February 2013 of peak magnitude close to 2,000 - 3,000 megalitres per day at Murchison
- baseflows in October, November and February 2013, which were managed to complement natural flows and support the on-going recovery of river-dependent native animals and plants.

Monitoring and evaluation

Monitoring the ecological response to environmental water delivered in the lower Goulburn River during 2012-13 focused on:

- water quality, nutrient levels, in-stream respiration and primary productivity, benthic algae, benthic invertebrate communities and zooplankton
- movement of organic matter (biotic dispersal)
- bank vegetation
- physical habitat for fish and macroinvertebrates.

Ecosystem responses

Primary productivity and movement of organic matter

It was expected that environmental water would increase the supply of organic matter to the river channel and would be an important driver of primary productivity. The environmental flow delivered in November 2012 washed large amounts of organic matter including snags and leaves from the river banks into the channel. The flow increased the amount of organic matter in the channel available for decomposers and provided an energy source for primary production in the river.

Macroinvertebrates

The fresh delivered in November 2012 appeared to temporarily reduce the abundance of macroinvertebrates, in particular those that are weak swimmers. However, macroinvertebrate abundance recovered quickly (in a matter of weeks) after the fresh.

Vegetation

Flows in the Goulburn River are far more variable than in Broken Creek, and accordingly the river supports different vegetation types to those found on the banks of Broken Creek. While the delivery of environmental water increased the variability of flow in the Goulburn River, the effects on vegetation in the River were minimal.

Fish and macroinvertebrate habitat

Environmental water delivered as baseflows increased the availability of slow-moving water ('slackwater'), which is important habitat for larval fish, invertebrate and macrophyte species, and deepwater habitat, which is important for large-bodied fish. Other habitats provided by environmental flows included large wood associated with tree roots, and woody debris at the toe of bank that would not otherwise have been underwater.

Implications for Commonwealth environmental water use

- Reductions in macroinvertebrate abundance during freshes are likely to be temporary and populations appear to recover or return quickly after recession.
- Response of bank vegetation to enhanced environmental flows will be greater in Broken Creek than Goulburn River, due to a history of stable baseflows in Broken Creek and hence the encroachment of terrestrial vegetation on the river bank.
- Freshes of short duration may lead to the shading of benthic algal communities already present, but not allow sufficient time for algae to colonise newly inundated habitat along the margins – this should be considered when planning for freshes.
- Managing for sediment smothering is difficult given what we know about its relation to flow and the types of flow we can provide

Broken Creek

Commonwealth environmental watering

Commonwealth environmental water contributed most of the flow in the lower Broken Creek in 2012-13, with a total of a total of 41,230 megalitres delivered between September 2012 and May 2013. Environmental water was managed to sustain flows of between 200 - 300 megalitres per day at Rices Weir for much of the monitoring period (October 2012 to February 2013).

With the exception of natural freshes in October and January, environmental water represented the bulk of streamflow. The expected outcomes of environmental watering in lower Broken Creek included improved fish movement through the fishway at Rices Weir and maintaining dissolved oxygen levels to support fish condition, movement and reproduction.

Monitoring and evaluation

Monitoring the ecological response to environmental water delivered in lower Broken Creek focused on:

- water quality, primary production and respiration
- dissolved oxygen (DO) levels
- fish movement within weir pools in relation to DO levels
- fish movement through fishways in response to flows.

Ecosystem Responses

Dissolved oxygen

This study provides strong evidence that environmental flows maintained moderate to high DO concentrations in lower Broken Creek weir pools. A number of the conditions that created hypoxic conditions in past years were again present in 2012-13, including similar carbon loads and sediment oxygen demands. However, flows in lower Broken Creek were higher than in previous years, due to the delivery of Commonwealth environmental water. Our modeling indicates that, in the absence of Commonwealth environmental water in 2012-13, DO levels in the weir would have been dangerously low for extended periods, and well below the ANZECC water quality guidelines for aquatic ecosystems.

Further, environmental water increased the velocity of flow in lower Broken Creek, which prevented any substantial temperature stratification. Temperature stratification was therefore only ever transient (lasting less than 4 hours) during 2012-13 and was not severe enough to restrict fish movement.

Dissolved oxygen and fish movement

Dissolved oxygen concentration in lower Broken Creek did not reach dangerous levels, and was not a driver of fish movement. Nonetheless the study has provided valuable baseline information on fish movement responses to small, short-term changes in DO and flows.

In particular, short-term oxygen depletion to about 2.5-3 milligrams per litre in lower Broken Creek did not appear to affect fish movement. If fish are exposed to more extensive periods of DO depletion, however, their responses are likely to differ.

Fish movement through fishways

Golden Perch

Increasing numbers of golden perch moved through fishways as discharge increased, and as the season progressed. These results are consistent with the PIT tag movement data, and data from previous investigations of golden perch. The results suggest that with an increase in average flow from 250 megalitres per day during September to 500 megalitres per day, there is a threefold increase in the number of golden perch passing through the fishways. Further, if high discharges (500 megalitres per day) are delivered in January, then the number of fish passing through the fishway will increase considerably more than this.

Murray cod

Murray cod showed a decline in fishway usage as water temperatures increased from September to March. Unfortunately, there were no high flow events during the summer trapping months that could have helped identify if temperature was really a 'driver' for Murray cod movement through fishways. However, the PIT tag data suggests that Murray cod will move during the warmer months if flooding is occurring. It is difficult to separate the effect of temperature and timing within the season (for Murray cod) given that there was a degree of collinearity between the two.

Implications for Commonwealth environmental water use

- Environmental water is highly effective in mitigating the risk of DO depletion in Broken Creek
- Native fish (namely golden perch and Murray cod), respond to a rising hydrograph by increasing their movement. A tailored environmental flow pattern may aid native fish movement - important for feeding, obtaining habitat, spawning and for colonisation/dispersal activities.

- Understanding the thresholds at which stress responses are elicited remains an important area for future research given hypoxic events have become increasingly frequent in waterways in the Murray-Darling Basin such as Broken Creek.
- These results have important implications for the provision of environmental flows designed to facilitate movement of Murray Cod and Golden Perch. Delivery of environmental water to lower Broken Creek that includes variations in flow may be effective in promoting movement of fish related to feeding, reproduction, dispersal
- High flows are likely to be effective in stimulating Golden Perch movement and, if lateral connectivity to the floodplain is avoided, the carp risk appears minimal.

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1 INTRODUCTION

The lower Goulburn River and Broken Creek monitoring program examines ecological responses to environmental flows over the 2012-13 watering year. This is the Stage 2 and final report. There are eight components to this monitoring program with four in the Goulburn River and four in lower Broken Creek. The Goulburn River components address:

1. Monitoring the contribution of environmental flows to nutrient levels, instream respiration and productivity, benthic algae, benthic invertebrate communities and zooplankton;
2. Monitoring the contribution of environmental flows to organic matter retention;
3. Examination of the cumulative effect of environmental flow releases on bank vegetation. This will be based on a comparison of bank vegetation in the Goulburn River with control sites on Broken Creek; and
4. Evaluation of the contribution of environmental flow releases to maintaining physical habitat for fish and macroinvertebrates, and in particular:
   i. inundated area of snags within the photic zone;
   ii. slackwater habitats;
   iii. deepwater habitats;
   iv. mobilisation of fine sediment.

   A model-based approach was used to evaluate the contribution of environmental flows over the entire 2012-13 season.

These four components address the following evaluation questions, respectively:

- What has been the contribution of environmental flow releases to riverine productivity in the lower Goulburn River (Section 3)?
- What has been the contribution of environmental flow releases to organic matter retention (Section 4)?
- Have environmental flows promoted flood tolerant vegetation on the stream bank in the lower Goulburn River (Section 5)?
- Have environmental flows improved physical habitat conditions for fish and macroinvertebrates? (Section 6)

The Broken Creek monitoring program addresses:

1. Water quality, primary production and respiration measurements;
2. Hydrodynamic data on mixing of dissolved oxygen (DO);
3. Observations of fish movement within weir pools in relation to DO levels; and
4. Observation of fish movements through fishways in response to flows.

These three components address the following four evaluation questions:

- Have environmental flows contributed to the maintenance of DO levels in lower Broken Creek weir pools? (Section 7)
- How is nutrient status and DO concentrations affected by flow manipulations? (Section 8)
- How is fish movement or activity affected by changes in DO and environmental flows in lower Broken Creek? (Section 9)
- Does fish movement through fishways increase with increasing river discharge? (Section 10).

2 SITES AND ENVIRONMENTAL WATER DELIVERY

2.1 Lower Goulburn River

Figure 2.1: Location of sites used in the monitoring program in the lower Goulburn River (modified from SKM, 2007)

The monitoring program uses four sites on the lower Goulburn River downstream of Nagambie, at Moss Road, Darcy Track, Loch Garry and McCoys Bridge (Figure 2.1). All four sites are used for monitoring effects on bank vegetation and are the same sites used for the VEFMAP (Victorian Environmental Flow Monitoring and Assessment Program). The Darcy Track and McCoys Bridge sites are used for monitoring the other three components for the CEWO Goulburn River program.

Commonwealth environment water was used to deliver environmental flow events in mid October 2012, mid November 2012 and early January 2013 (Figure 2.2). It is important to note that prior to commencement of this monitoring program, the Goulburn River experienced relatively high natural flows in winter and spring 2012, and very high summer and winter natural flows in preceding years (Figure 2.3).

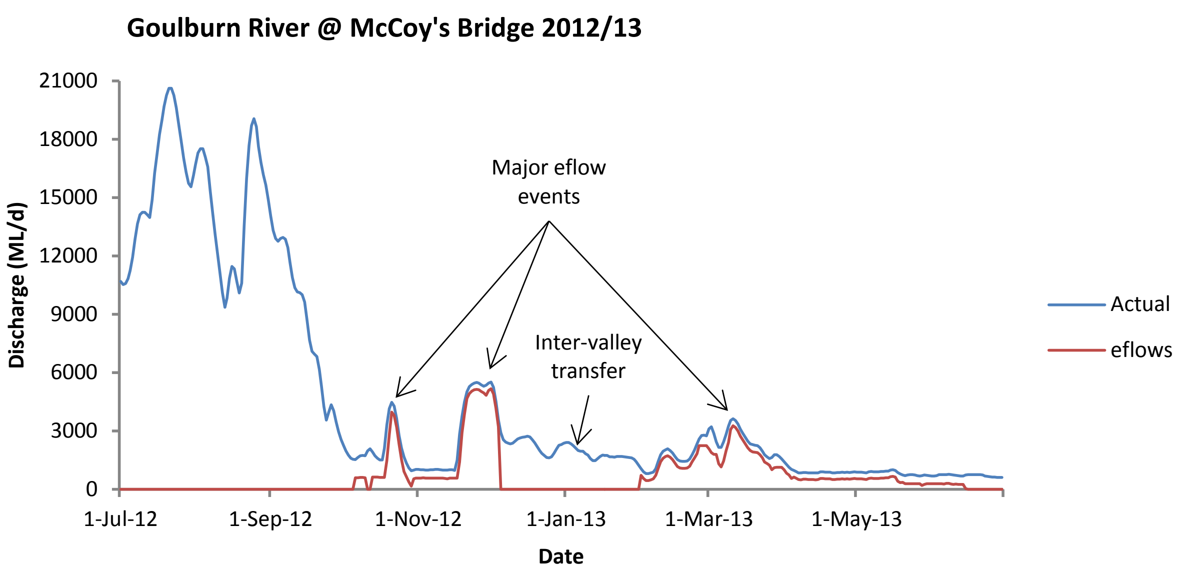

Figure 2.2: Flows in the Goulburn River from July 2012 to June 2013, showing the environmental flow contribution. Flows are shown in ML/day and measured at McCoys Bridge (data provided by Goulburn-Murray Water).

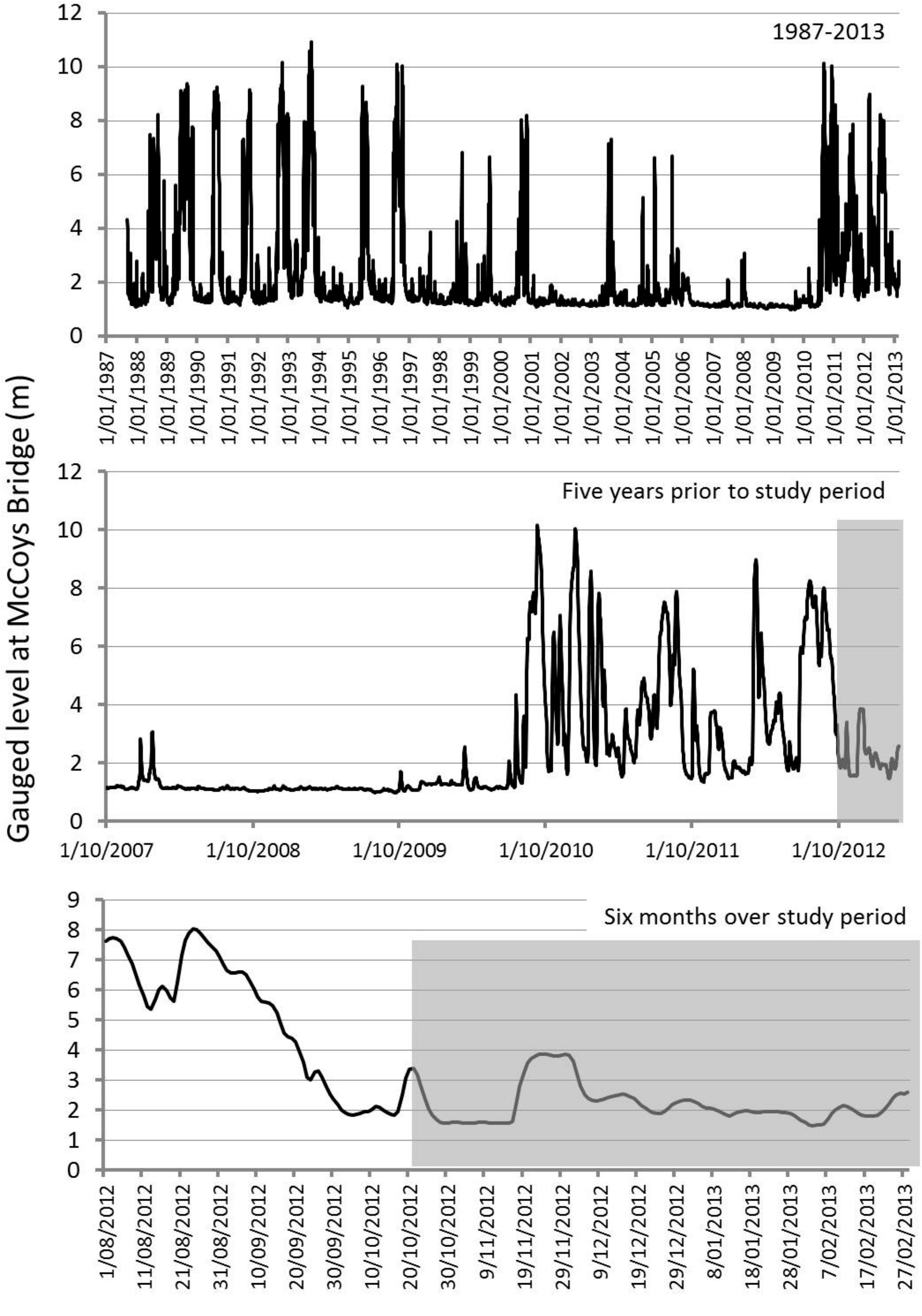

Figure 2.3: Antecedent flows prior to the sampling period (indicated by the grey shaded box). Flows are gauged levels at McCoys Bridge (Murray Darling Basin Authority, Site ID 405232C, Goulburn River @ McCoys Bridge, -36,17770454, 145.1190535)

2.2 Broken Creek

Broken Creek was, historically, an ephemeral creek. A series of weirs along lower Broken Creek and the regulated diversion of flow into Broken Creek allowed permanent irrigation to be established along the lower reaches (Figure 2.4). With a catchment of approximately 3300 km^2, Broken Creek flows west, from the Warby Ranges and the foothills around Dookie, through northern Victoria to meet the River Murray at the downstream end of the Barmah Forest (Figure 2.5). Mean daily discharge in the creek is approximately 230 ML/d (range: zero to 2139 ML/d). Ten low level weirs control flows along Broken Creek between Numurkah and Barmah, these were constructed to regulate flows for agricultural purposes. Each of the ten low-level weirs has a vertical-slot fishway to pass fish.

Broken Creek is surrounded by agricultural land, however the stream bank is lined with river red gum (*Eucalyptus camaldulensis*), and grey box (*Eucalyptus microcarpa*). Instream habitat consists of interspersed emergent cumbunghi (*Typha* spp.), spike rush (*Eleocharis* spp.), common reed (*Phragmites australis*), whilst large and small woody debris are frequently encountered.

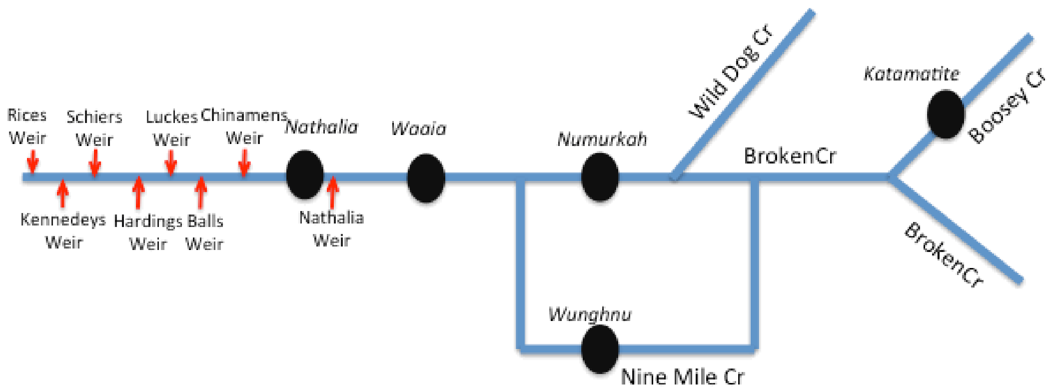

Figure 2.4 Schematic map of Broken Creek system

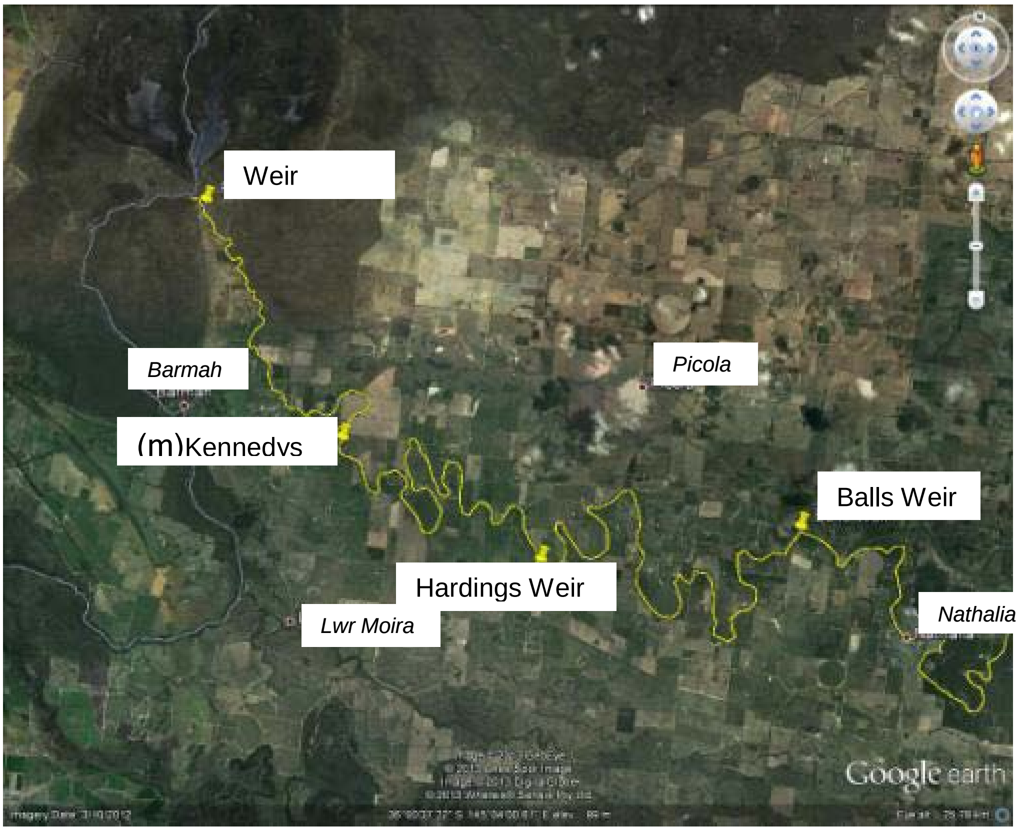

Figure 2.5 Lower Broken Creek and four of its weir pools downstream of Nathalia.

The final weir on lower Broken Creek, Rices Weir, has been beset by water quality issues, especially low dissolved oxygen. Low dissolved oxygen levels were thought to have led to a major fish kill in 2002. Environmental water has been delivered to the lower Broken Creek during the summer of 2012-2013 in an attempt to improve water quality and fish habitat (Figure 2.6). Commonwealth environmental water contributes most of the flow in the lower Broken Creek between October and March. This study reports on monitoring of water quality and fish movement at some of the weirs in the lower Broken Creek and the effectiveness of Commonwealth environmental water delivery.

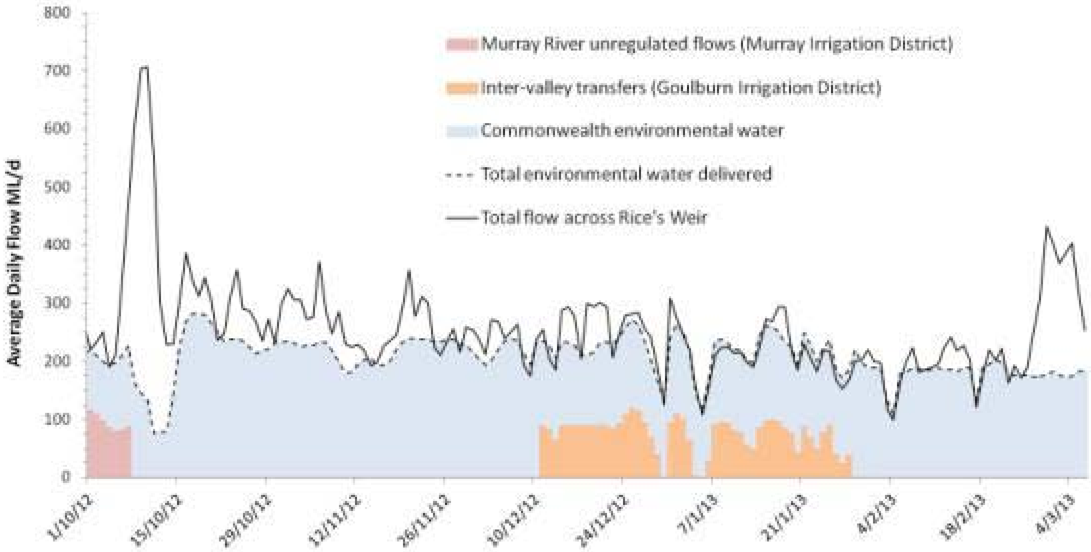

Figure 2.6: Breakdown of water sources contributing to flow across Rices Weir, Broken Creek, between October 2012 and March 2013. Source: G. Earl, 2013, GBCMA

3 RIVERINE PRODUCTIVITY

3.1 Evaluation question and expected outcomes

This section addresses the evaluation question: What has been the contribution of environmental flow releases to riverine productivity in the lower Goulburn River? The expected outcome was that environmental flows delivered as freshes would stimulate a temporary increase in primary productivity due to the mobilisation of organic matter from the river bank.

3.2 Methods

Sampling for this component of the project took place from late October 2012 until early March 2013. As a result, the most informative data on the effects of environmental flows occurs around the flow peak delivered in mid November 2012, where there is data before and after the flow (Figure 2.2). Observation during these environmental watering events can be compared with observations during natural flow peaks in mid and late December 2012. These natural peaks were of a similar magnitude to environmental watering events in late January and onwards into February 2013 but smaller than the environmental watering in November.

Sampling took place at two 50 m reaches on the lower Goulburn River at Darcy Track (hereafter Darcy) and McCoys Bridge (hereafter McCoys) (Figure 2.1). Those reaches are typical of this part of the river, with a strongly U-shaped channel with steep banks and extensive native riparian vegetation extending to the top of the banks. The active channel has some smaller woody vegetation within it, but also extensive areas of bare ground and fallen woody debris (snags) (Figure 3.1). In order to effectively illustrate changes in flows over the study period, fixed automatic cameras were placed at each site pointing downstream. Photos were taken at half hourly intervals over the study period.

It is important to note that differences between sites are not of interest in the context of this report, rather the sites are used as replicates to assess the effects of environmental flows on the indicators of interest. As a result, the emphasis in describing results will be on changes which may be correlated with changes in flow within sites, not on broader questions around patterns between sites or along the river.

Figure 3.1: Study reaches on the lower Goulburn River at Darcy Track (top) and McCoys Bridge (bottom), photographed on the 13 October 2012.

Sites were sampled from late October 2012 to late February 2013 for a number of key variables (Figure 3.2). Sampling was generally biweekly (see details for each variable below) but there was no sampling over the Christmas/New Year period (Figure 3.2).

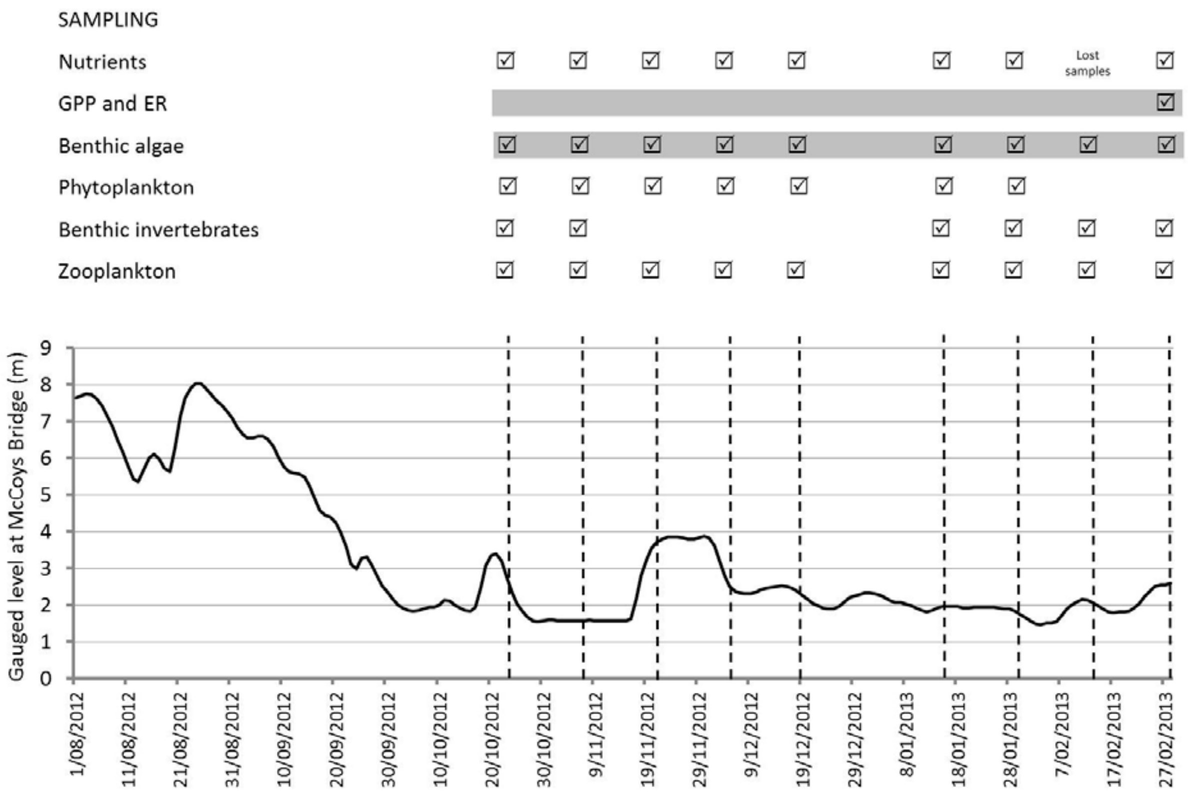

Figure 3.2: Timing of sampling (dotted vertical lines) for measures of riverine productivity at Darcy Track and McCoys Bridge on the lower Goulburn River relative to river height. Grey shading indicates continuous measurement. Boxed ticks indicate when sampling occurred. GPP = gross primary production, ER= ecosystem respiration.

Nutrient concentrations

Duplicate water samples were taken biweekly 2 m from the water's edge at 1 m depth at three locations within each site (Figure 3.2). These were placed on ice and returned to Monash University Water Studies Centre (NATA Accredited ISO/IEC 17025) for analysis for total nitrogen [persulphate digestion method and automated cadmium reduction method (APHA, 2013)], nitrate/nitrite [automated cadmium reduction method (APHA, 2013)], total phosphorus [persulphate digestion method and the automated ascorbic acid reduction method (APHA, 2013)], dissolved reactive phosphorus [after filtering in the field through a 45 µm Whatman cellulose-acetate filter, automated ascorbic acid reduction method (APHA 2013)] and ammonia [automated phenate method (APHA, 2013)] concentrations.

In-stream respiration and productivity

Rates of primary production and respiration were estimated by continuously recording dissolved oxygen concentrations (DO), temperature and ambient light (as photosynthetically active radiation [PAR]) at the two sites between 21 November 2012 and 27 February 2013

(Figure 3.2). Primary production measures the amount of carbon fixed by photosynthetic organisms (algae, phytoplankton, macrophytes) during daylight, and respiration measures the amount of carbon respired by consumers (bacteria, fungi, invertebrates, fish). Very high concentrations of oxygen occur when levels of photosynthesis are very high, associated with algal 'blooms'. Very low levels of oxygen can be associated with high levels of decomposition due to high inputs of organic matter. Oxygen depletion due to the decomposition of material from collapsing algal blooms or high levels of dissolved organic carbon ('blackwater' events) can result in deaths of fish or invertebrates due to lack of oxygen.

Data sondes (Hydrolab 3, Hydrolab Inc., USA) were used to record DO and water temperature every 10 minutes over the study period. At each site one data sonde was placed at a location where there were no bank features to generate backwaters or eddies at height of 1 m above the stream bed. As a result of fluctuations in water levels the sondes varied between 0.6-3.0 m below the surface (average 1.1 m), but were consistently 1 m from the stream bed. Every four weeks data were downloaded and the sondes were recalibrated. PAR data were recorded every 10 minutes on a separate logger placed in direct sunlight. As the channel was wide (>10 m) at both locations, the metabolic parameters, gross primary production (GPP) and ecosystem respiration (ER), plus the reaeration rate were calculated for each day as described by Atkinson et al. (2008). The well-mixed nature of the channel makes these methods appropriate (see Results). Daily GPP and ER for each reach were calculated, as well as the ratio of GPP:ER.

Shading effects at the sites is less important because we were interested in changes in GPP and ER through time associated with changes in flow regime and not in comparing sites. It is also likely that bank shading will be relatively small in a wide channel like the lower Goulburn River and because rates of photosynthesis saturate at relative low levels, shading by the kind of sparse vegetation in these systems is unlikely to affect in-channel processes. In narrower channels with higher banks this assumption may not be valid.

Benthic algae (biofilm)

Benthic algae were assessed using colonisation blocks of red gum sampled biweekly (Figure 3.2). The biomass and diversity of biofilms was assessed in the river reaches before and after environmental flows. Algal colonisation arrays were deployed at the mid-point of each reach (Figure 3.3). Each array consisted of 30, 5 cm x 5 cm blocks of river red gum wood, arranged in ten rows of three blocks. Arrays were placed 5 cm below water level when the gauge at McCoys Bridge reads 1.5 m (targeting the area where light will fluctuate), 50 vertical centimetres below water level at the 1.5 m point (where light is likely to be limited during environmental flows) and 50 vertical centimetres above the water level at the 1.5 m point (areas which will be intermittently wet during environmental flows). Arrays were sampled by removing one 'column' of three blocks from each of the three arrays at each location on a sampling occasion (Figure 3.2). Each array had an associated depth logger to continuously measure the amount of water over each array, and a light/temperature logger. Arrays were installed on 10 October 2012 and sampled bi-weekly thereafter, except 31 December 2012 (Figure 3.2).

Upon sampling, blocks were immediately placed on ice and in the dark for return to the laboratory, where they were frozen. Blocks were processed for algae approximately monthly.

The biofilm was scrubbed in the field from each red gum block into 1000 mL of distilled water using a soft nailbrush and then placed on ice for return to the laboratory. The sample was thoroughly homogenised and then 200 mL was filtered through a GC-50 0.5 µm filter for determination of chlorophyll-a. The filter paper was extracted in 10 mL of acetone in the dark at 4^{o} C for 12 hours. At the end of this time the acetone was filtered through a GF-C filter to remove any sediment and then chlorophyll-a was measured spectrophotometrically (Shimadzu UV-200, Shimadzu, Japan) using standard methods (APHA 2013).

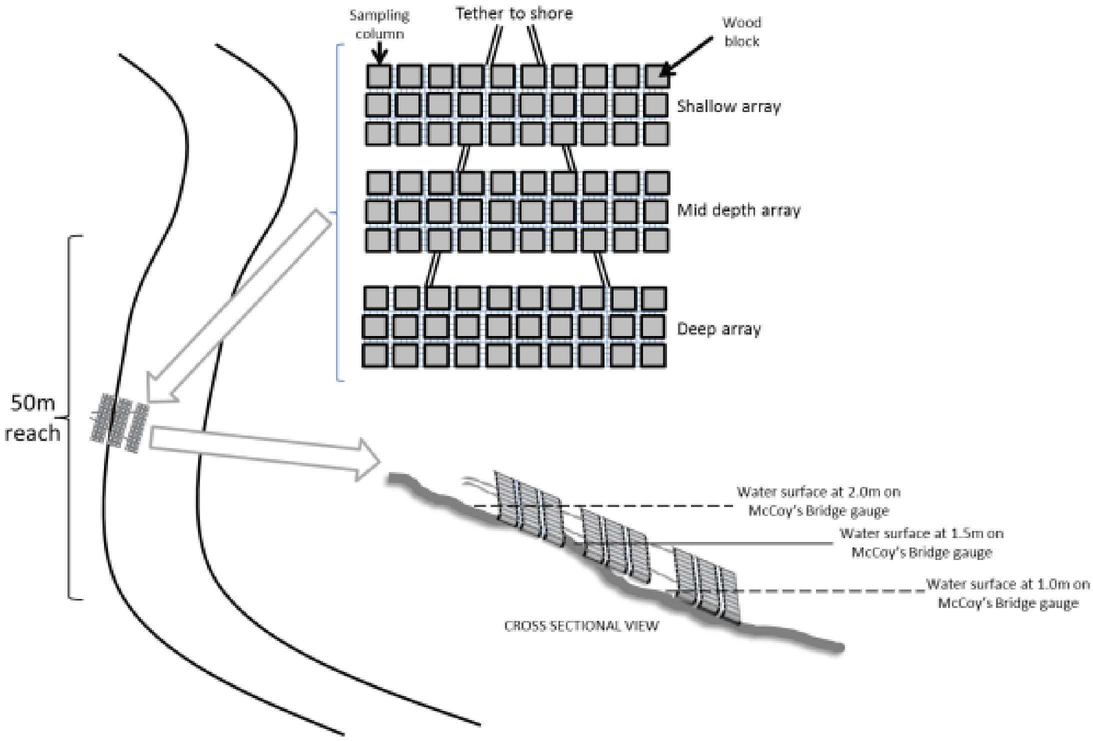

Figure 3.3: Assessment of benthic algal response to environmental flows. Overall layout per site (left panel), the arrangement of wood blocks within an array (top right) and a cross section showing deployment of arrays by depth (bottom right).

A further 200 mL was filtered through a pre-ashed and pre-weighed GC-50 0.5 µm filter papers. That filter paper was dried at 80°C for 24 hours, weighed, combusted for four hours at 500°C and reweighed. All samples were weighed to four decimal places and converted to dry weight (DW) and ash free dry weight/organic biomass (AFDW). Percent organic matter was calculated as the proportion of AFDW to DW and converted to a percentage to standardise across sites and dates.

A further 50 mL sample was taken and preserved with Lugol's iodine. This was allowed to settle and then a concentrated 1 mL sample was taken from the bottom of the vessel and made into a slide for microscopic identification of algae.

Chlorophyll and AFDW data were averaged for the three wood blocks at each depth within the two sampling locations within each reach. For each depth therefore, there were two values per reach for each date, yielding an average and standard error value.

Phytoplankton

Phytoplankton biomass and composition was measured bi-weekly. Duplicate 200 mL samples were taken at 5 haphazard locations along each site at 10 cm depth 1 m from shore. Samples were pooled to comprise two 1 L samples and stored on ice for return to the laboratory. In the laboratory for each sample one 250 mL sub-sample was taken and settled for 24 hours in the dark before being preserved in Lugol's iodine and used for taxonomic analysis to identify dominant algal groups. A further 250 mL was filtered through a pre-ashed and pre-weighed GC-50 0.5 µm filter paper. That filter paper was dried at 80°C for 24 hours, weighed, combusted for four hours at 500°C and reweighed. All samples were weighed to four decimal places and converted to dry weight (DW) and ash free dry weight/organic biomass (AFDW). Percent organic matter was calculated as the proportion of AFDW to DW and converted to a percentage to standardise across sites and dates. A third 250 mL was filtered through a GC-50 0.5 µm filter for determination of chlorophyll-a. The filter paper was extracted in 10 mL of acetone in the dark at 4° C for 12 hours. At the end of this time the acetone was filtered through a GF-C filter to remove any sediment and then chlorophyll-a was measured spectrophotometrically (Shimadzu UV-200, Shimadzu, Japan) using standard methods (APHA 2013).

Each sampling date at each reach provided two measurements of phytoplankton chlorophyll, AFDW and species composition. For chlorophyll and AFDW data were averaged for each reach and sampling date. Due to an equipment failure, phytoplankton were not sampled for the last two sampling occasions.

Benthic invertebrate communities

Invertebrate community analysis was carried out through biweekly sampling of marginal habitat. Three sweep net samples were taken, each over 10 m of stream bank with a standard 250 micron mesh net. Five additional samples were taken from snag habitat using standard ‘snag bags’ with 250 µm mesh. All samples were preserved in ethanol for return to the laboratory and subsequent analysis. All animals were identified to family and counted. On three occasions (Figure 3.2) high flows made it too dangerous to undertake benthic invertebrate sampling.

For analysis, animals were identified to family level, or genus where possible. All individuals in samples were counted. For statistical analysis data from the five snag samples and three sweep samples were averaged for each reach on each sampling occasion. Average abundances in sweep and snag samples were calculated. Data for snags and sweeps were analysed separately in a multivariate analysis using Primer 6.13 (Plymouth Marine Laboratories). Data were log(X+1) transformed and a Bray-Curtis similarity analysis across all samples was calculated. This was represented as a non-metric multi-dimensional scaling ordination. Taxa characterising each reach and each sampling occasion were determined using the SIMPER procedure. To determine whether there was any effect of flow on the two reaches analysis of similarities (ANOSIM) was used to test for differences between sampling dates. SIMPER was used to determine what taxa drove differences in communities between sampling dates.

Zooplankton

Zooplankton sampling was undertaken bi-weekly. Three 5 m plankton hauls were taken at 20 cm depth, using a mesh size of 100 µm. Samples were preserved in ethanol for return to the laboratory and subsequent analysis. All animals were identified to family and counted. For statistical analysis data from the three hauls were averaged for each reach on each sampling occasion. Average abundances in samples were calculated. Data were analysed in a multivariate analysis using Primer 6.13 (Plymouth Marine Laboratories). Data were log(X+1) transformed and a Bray-Curtis similarity analysis across all samples was calculated. This was represented as a non-metric multi-dimensional scaling ordination. Taxa which characterised each reach and each sampling occasion were determined using the SIMPER procedure. To determine whether there was any effect of flow on the two reaches analysis of similarities (ANOSIM) was used to test for differences between sampling dates. SIMPER was used to determine what taxa drove differences in communities between sampling dates.

3.3 Results

Water level varied substantially over the study period, from 0.80 m on the McCoys Bridge gauge to almost 4 m on the gauge. Sampling in early November occurred during the lowest flows (Figure 3.4 A,B) with flows peaking in mid November (Figure 3.4 C,D) with smaller natural flows in early January (Figure 3.4 E,F).

Figure 3.4: Photos from fixed photo points at Darcy (left column) and McCoys (right column) at the following gauge levels at McCoys Bridge; A + B 26/10/2012 1.86 m, C + D 22/11/2012 3.84 m, E + F 4/1/2013 2.20 m.

Nutrient concentrations

Nutrient concentrations were measured in terms of nitrate/nitrite (hereafter nitrate), ammonia, total dissolved phosphorus (TDP) and dissolved reactive phosphorus (DRP). Nitrate, ammonia, TDP and DRP levels were periodically higher than the water quality guidelines for lowland rivers in this area (total nitrogen 0.422 mg/L, nitrate = 0.004 mg/L; ammonia = 0.030 mg/L; TDP=0.032 mg/L, DRP = 0.005 mg/L) (ANZECC 2000) but there were no clear patterns with flow or provision of environmental flows (Figure 3.5). There was some evidence of a slight increase in phosphorus levels after the November environmental flows that may be consistent with mobilisation of sediments from the bed or banks, but this effect was not persistent. There was no clear evidence for consistent patterns in nitrate or ammonia in response to flow fluctuations. Samples for mid-February were lost due to a mishap in the laboratory.

In-stream respiration and productivity

The data from the sondes were of high quality and showed clear day-night cycling in dissolved oxygen concentrations. Diel curves were successfully fitted (correlation coefficient >0.9) using the Atkinson method for 96% of the days where there were data. Curves could not be fitted for the remaining occasions and were ignored for analysis. This problem occurs when objects (such as leaves) temporarily lodge on the sensor. Several periods of data were lost due to equipment failures, including the loss of all the data for December at the McCoys site when the datasonde unit failed. Other interruptions to data were caused by vandalism, when the sondes were pulled from the water (mid October/early November; McCoys) and a battery failure (December; Darcy). Despite this, there were data available before and after the main environmental flow in mid November (Figure 3.6). At no stage over the study did the loggers record very low oxygen levels (<20%) that might have been indicative of blackwater events (i.e. an event where very high levels of dissolved organic carbon lead to large depletions in water column oxygen).

Values for both gross primary productivity (GPP) and ecosystem respiration (ER) were typical of those found in large slow flowing rivers world-wide and in Australia (Watts et al. 2013). The ratio between GPP and ER (sometimes referred to as the P/R ratio) indicates the balance between primary production rates and ecosystem respiration rates, with a value <1 showing that more organic carbon is being consumed in the study reach than is being produced by photosynthesis (and therefore that carbon must be entering the reach from outside it in some form; either in the flow or from leaf litter etc entering the channel). Both sites generally had GPP:ER of <1 with the exception of the McCoys site (data was not available for the Darcy site at this time) after a period of prolonged low flow in November 2012, when GPP was particularly high and exceeded rates of ER.

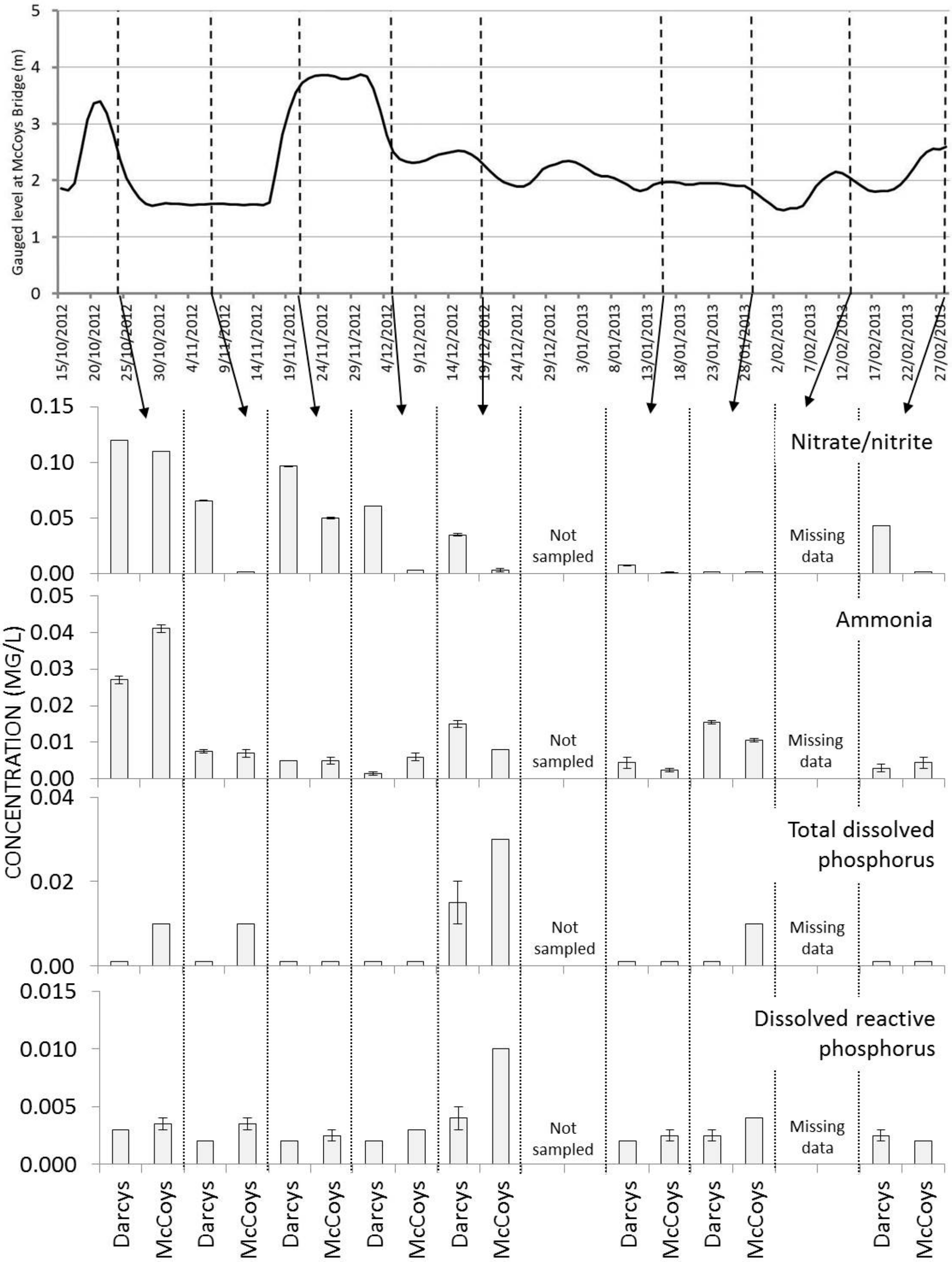

Figure 3.5: Nutrient concentrations (in mg/L) in the water column for duplicate samples (shown as means with standard error bars) taken from the lower Goulburn River at two reaches (Darcy and McCoys) at different river levels (top panel). Dotted lines and dates indicate dates of sampling.

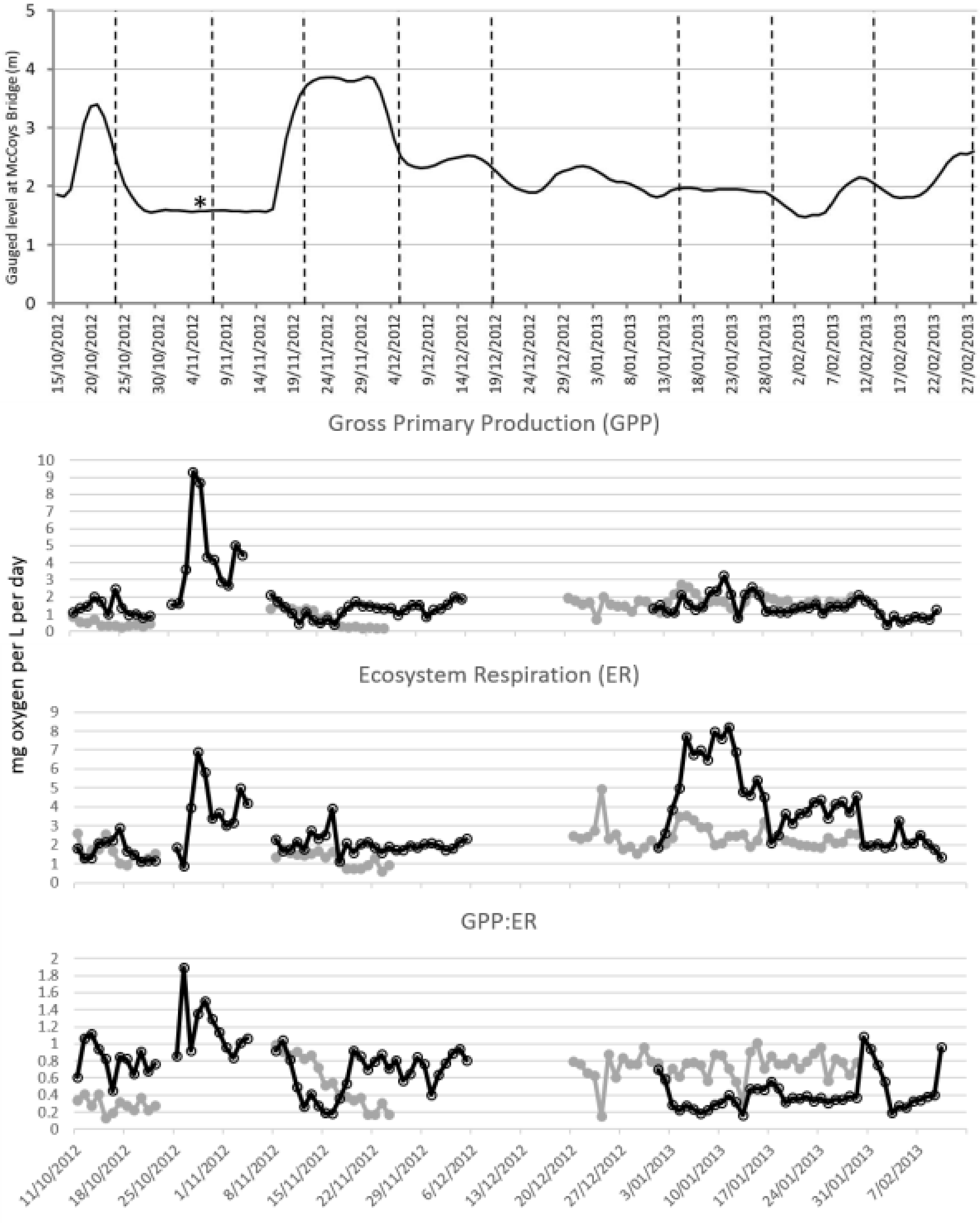

Figure 3.6: Rates of gross primary production, ecosystem respiration and the ratio between them taken at two reaches (Darcy [grey line] and McCoys [black line]) from the lower Goulburn River at different river levels (top panel).

There was evidence that GPP, ER and GPP:ER were reduced after the main environmental flow in November (Figure 3.6) and this was particularly striking for GPP:ER. The reaches became more heterotrophic (dominated by carbon entering the reach from outside, rather than by photosynthesis) after the environmental flow commenced. This effect was reduced as the flow persisted, suggesting that autotrophic productivity was able to ‘rebound’ after the effects of flow, potentially via algal colonisation of newly inundated banks or from phytoplankton recovery after initial dilution or washout.

Benthic algae (biofilm)

The reach at McCoys consistently had larger amounts of biofilm than the Darcy reach, although this difference was most pronounced early in sampling. At both sites there tended to be less algae in deeper water. During higher flows in the river (21/11/2012 onwards) the amount of algae reduced, particularly in deeper water. This was most pronounced for the chlorophyll-a data, suggesting that the algae may have been becoming moribund. Highest levels of algal biofilms were recorded when the river was at relatively low and stable flows in early November. The period of sampling reported here described patterns in benthic algae over two periods of heightened flow for chlorophyll-a (Figure 3.7) and ash free dry mass of biomass (Figure 3.8).

Algal communities were general dominated by green algae (particularly *Closterium*) and diatoms (predominantly *Navicula* and *Achnanthes*) (Table 3.1). Blue green algae were present, but in lesser amounts and never dominated the communities. Filamentous green algae predominated in the early samplings, but were much reduced after the November environmental flow, where upon diatoms became increasingly abundant (particularly *Navicula*). There was no clear pattern of composition by water depth, or differences between reaches, although there was some evidence of seasonal succession.

Table 3.1: Dominant biofilm taxa from scrubbings of wooden blocks from two reaches on the lower Goulburn River sampled at three depths between Oct 2012 and Feb 2013.

CHLOROPHYTA (green algae)	CYANOPHYTA (blue-green algae)
Bulbochaete spp.	*Anabaena* spp.
Closterium spp.	*Lyngbya* spp.
Coelastrum spp.	*Nodularia* spp.
Cosmarium spp.	*Oscillatoria* spp.
Draparnaldia spp.	*Phormidium* spp.
	Schizomeris spp.
FILAMENTOUS GREEN	BACILLARIOPHYTA (diatoms)
Mougoutia spp.	*Achnanthes* spp.
Oedogonium spp.	*Amphora* spp.
Pediastrum spp.	*Fragillaria* spp.
Pleurotaenium spp.	*Melosira* spp.
Scenedesmus spp.	*Navicula* spp.
Spirogyra spp.	*Nitzschia* spp.
	Pinnularia spp.

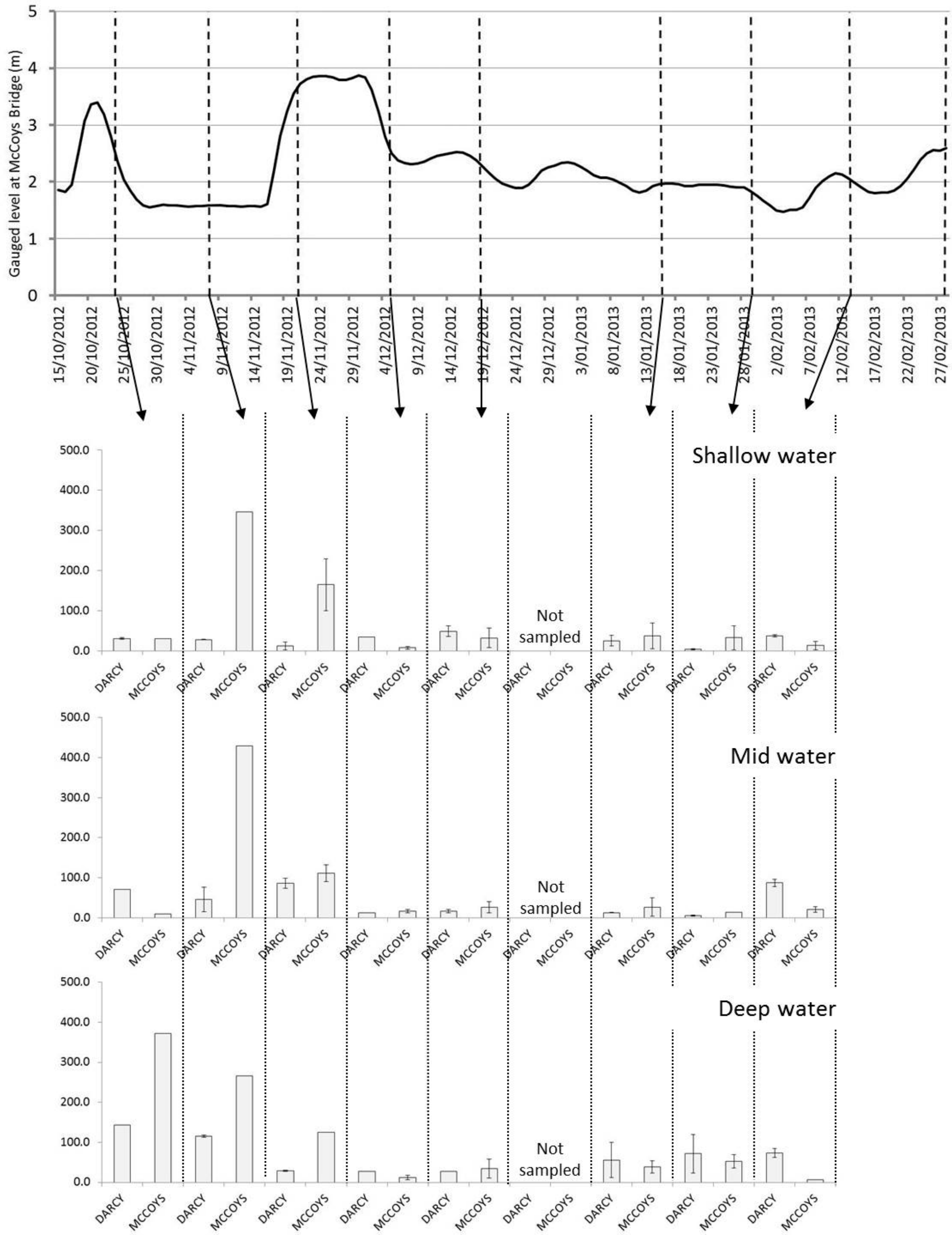

Figure 3.7: Responses of benthic algae (measured as chlorophyll-a) grown on colonisation blocks to flow (top panel) in the lower Goulburn River at two reaches (Darcy and McCoys). Dotted lines and dates indicate dates of sampling. Results are shown for blocks placed in shallow, medium and deep water (see Methods). Values are means for two locations in each reach with standard errors.

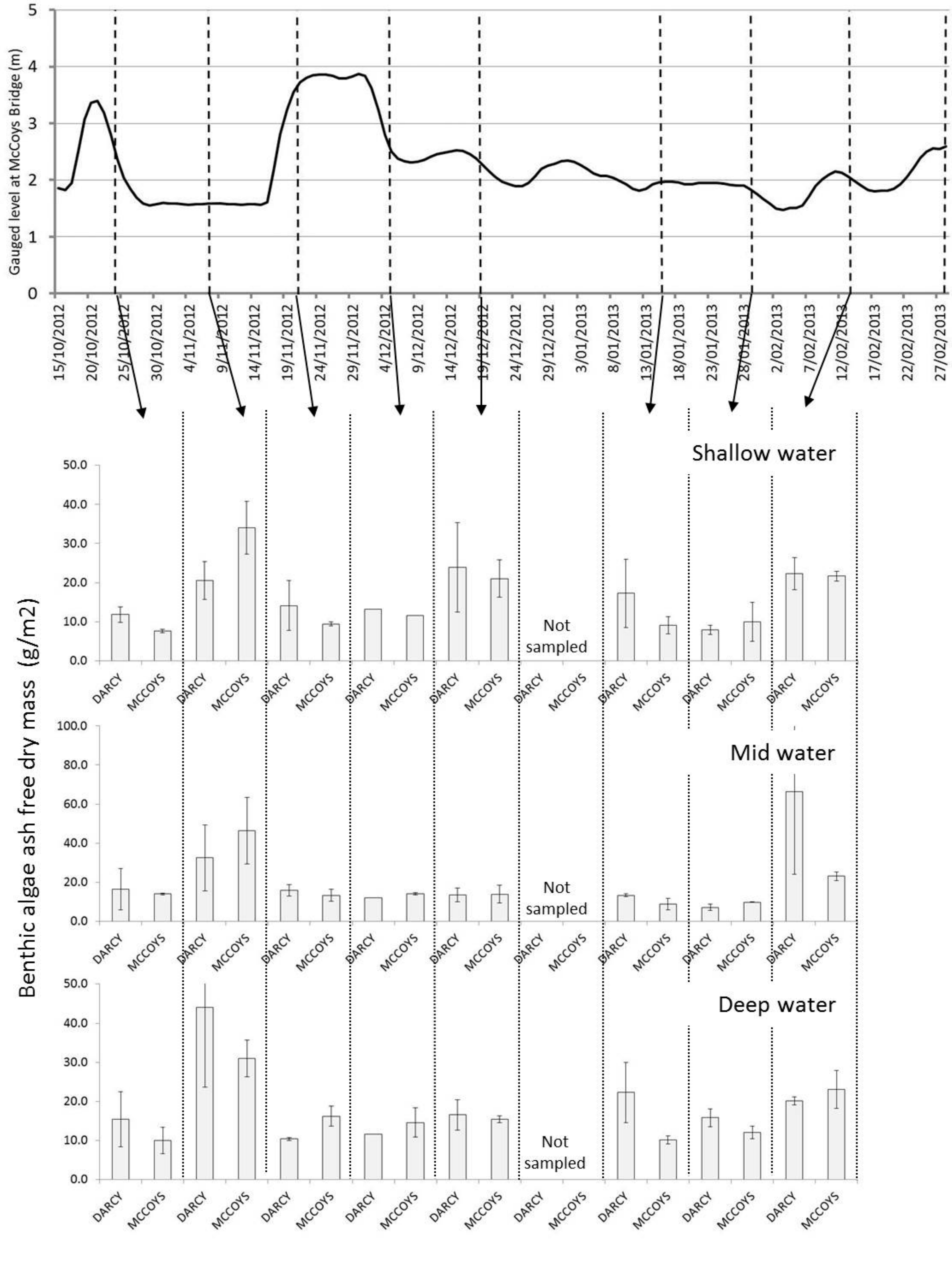

Figure 3.8: Responses of benthic algae (measured as ash free dry weight of biomass) grown on colonisation blocks to flow (top panel) in the lower Goulburn River at two reaches (Darcy and McCoys). Dotted lines and dates indicate dates of sampling. Results are shown for blocks placed in shallow, medium and deep water (see Methods). Values are means for two locations in each reach with standard errors.

Phytoplankton

Phytoplankton showed no clear patterns by site, or in relation to flow for either ash free dry mass or chlorophyll-a (Figure 3.9). It is important to note that the AFDW measure measures all organic matter in the water column, (seston + phytoplankton). Microscopic inspection of the slides suggested that only a small portion (<1%) of the biovolume were clearly identifiable algal cells. There was some evidence that AFDW was higher during the November environmental flow, consistent with organic matter being resuspended. Composition data was extremely variable, with samples generally dominated by the same taxa found in the benthic biofilms (compare Table 3.2 with Table 3.3) suggesting that these were largely sloughed cells in the water column rather than phytoplankton per se. This hypothesis is supported by the fact that phytoplankton biomass was high as flows increased in November (Figure 3.9).

Table 3.2: Dominant biofilm taxa identified from phytoplankton samples taken from two reaches of the lower Goulburn River between October 2012 and January 2013.

CHLOROPHYTA (green algae)	CYANOPHYTA (blue-green algae)
Bulbochaete spp.	*Anabaena* spp.
Chlorella spp.	*Lyngbya* spp.
Closterium spp.	*Nodularia* spp.
Coelastrum spp.	*Oscillatoria* spp.
Cosmarium spp.	*Phormidium* spp.
Draparnaldia spp.	
FILAMENTOUS GREEN	BACILLARIOPHYTA (diatoms)
Oedogonium spp.	*Achnanthes* spp.
Pleurotaenium spp.	*Amphora* spp.
Scenedesmus spp.	*Fragillaria* spp.
Spirogyra spp.	*Melosira* spp.
	Navicula spp.
	Nitzschia spp.
	Pinnularia spp.

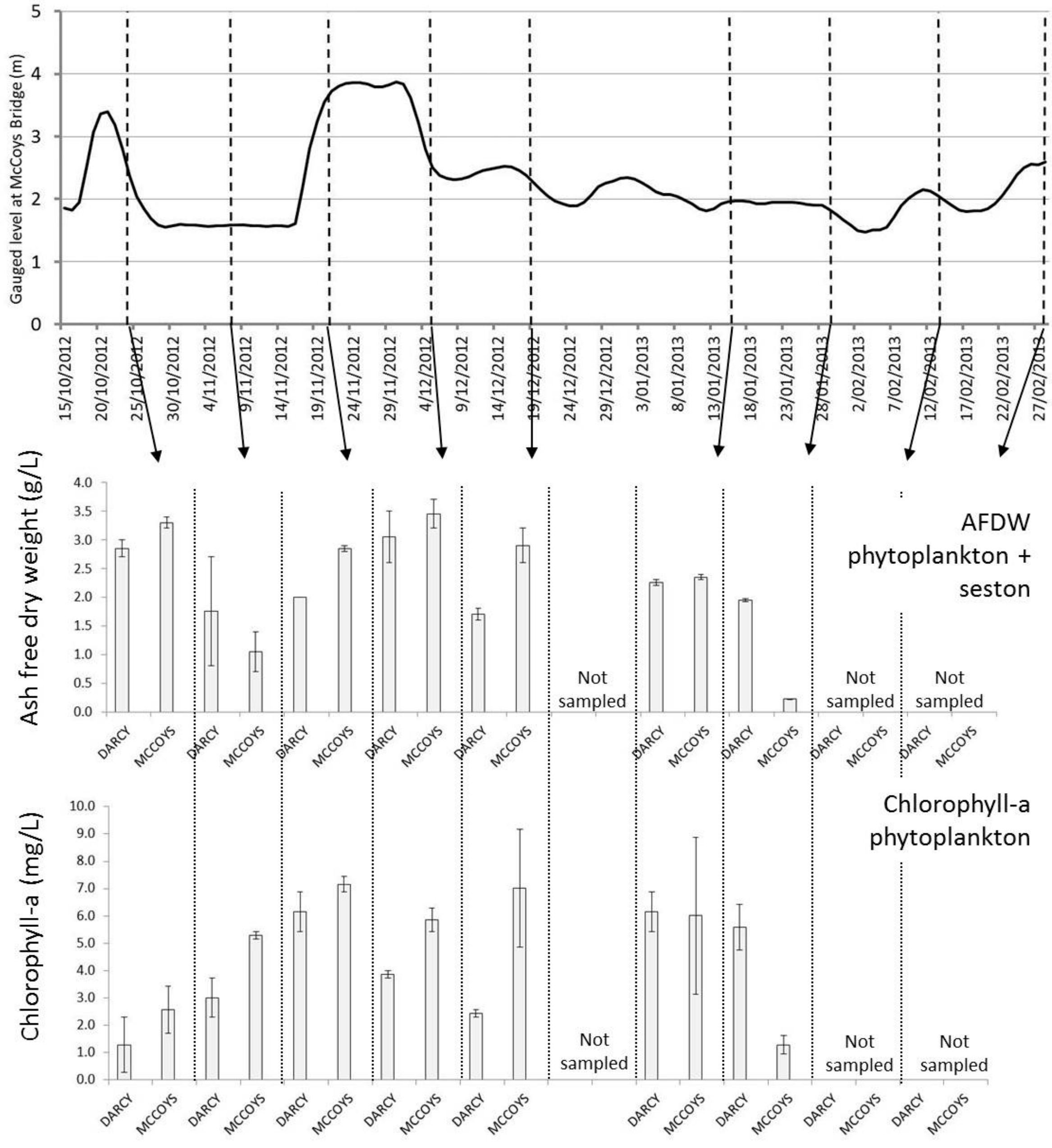

Figure 3.9: Responses of phytoplankton to flow (top pan), measured as ash free dry weight of biomass (includes seston and phytoplankton) (middle panel) and chlorophyll-a (bottom panel) collected from the water column in the lower Goulburn River at two reaches (Darcy and McCoys). Values are means of two samples per reach with standard errors. Dotted lines and dates indicate dates of sampling.

Benthic invertebrate communities

Invertebrate communities were highly variable in the snag samples from both reaches (Table 3.3). Abundances of animals on snags tended to be higher at the McCoys reach but there was no clear pattern for taxa richness (Figure 3.10).

Benthic communities were dominated numerically by the chironomids, nematodes, oligochaetes and elmid beetles caddis in the snag samples from both sites (Table 3.3). However there was very high variability between samples within sites (Darcy 36.9% similarity between samples, McCoys 32.7%). Samples from both sites were characterised by abundant chironomids and Ecnomidae caddis, but whereas the Darcy reach community also included abundant mayflies (Leptophelbiidae) and nematodes, the McCoys reach was characterised by higher abundance of elmid beetles, corixid bugs (Micronecta) and empid fly larvae.

Sampling through time revealed some relationships between flow and community structure across the sites (Figure 3.10). There was very low abundance of all taxa for samples taken on the rising limb of a high flow event in late November. It is not possible to determine whether this was due to a true change in the communities or to difficulties in sampling the habitats – it was dangerous to attempt to access snags in deeper water at those flow levels. Sampling was not carried out at the end of February because of the dangerous nature of the high flows and slippery banks. Comparing the 7/11/12 (relatively low flows) samples with those from 5/12/12 (immediately after a period of high flow) suggested that high flows resulted in a reduction in the abundance of chironomids, although numbers recovered over the following month. Leptophlebiid mayflies, elmid beetles, ecnomid caddis and empid fly larvae were all at lower numbers after the November high flow event, although numbers recovered quickly. In contrast *Micronecta* was more abundant after the high flows. Analysis of similarities found an effect of date of sampling (Global R = 0.173, p=0.001) even when differences between sites were taken into account. This result was driven by difference between the community on the 21/11/12 (at high flows) and 5/12/12 (immediately after high flows) and the communities sampled at all other times, which were not different from one another. This suggests an impact of the higher flows on community composition. In particular these sampling occasions were characterised by samples with no individuals, which strongly drove patterns. When these samples were removed patterns amongst sites and dates were more readily evident. The communities that were most different were those from the high flow and immediately high flow sampling dates (21/11/12 and 5/12/12) (Figure 3.11).

Table 3.3: Macroinvertebrate taxa identified from snag samples from two sites on the lower Goulburn River sampled between October 2012 and February 2013.

SITE	Sample Date	Ceratopogonidae	Chironomidae	Ecnomidae	Elmidae	Empididae	Griptopterygidae	Hydraenidae	Hydrophillidae	Hydroptilidae	Leptophlebiidae	*Micronecta* spp.	Nematoda	Nemertomorpha	Noteridae	Notonectidae	Notonemouridae	*Offadens* spp.	Oligochaeta	*Paratya* sp.	Simuliidae	Staphylinidae	*Tasmanocoenis* spp.	Tipulidae	Veliidae	TOTAL # taxa	TOTAL abundance
DARCY	24/10/2012		34	1	1	2								1												5	39
DARCY	24/10/2012		20	1	2	2																				4	25
DARCY	24/10/2012		1								1															2	2
DARCY	24/10/2012		1		1						1															3	3
DARCY	24/10/2012		3																							1	3
MCCOYS	24/10/2012		56		3	1								1												4	61
MCCOYS	24/10/2012		35		4						1															3	40
MCCOYS	24/10/2012		2								1															2	3
MCCOYS	24/10/2012		20	1	4						4															4	29
MCCOYS	24/10/2012		15																							1	15
DARCY	7/11/2012		1								1															2	2
DARCY	7/11/2012		12								1															2	13
DARCY	7/11/2012		22								1															2	23
DARCY	7/11/2012		10																							1	10
DARCY	7/11/2012		2		1						1															3	4
MCCOYS	7/11/2012		12	17	7	8																				4	44
MCCOYS	7/11/2012		2			1																				2	3
MCCOYS	7/11/2012		56	13	11	10												2								5	92
MCCOYS	7/11/2012		30	4	1						3															4	38
MCCOYS	7/11/2012		22	2	1	2																				4	27
MCCOYS	21/11/2012										2															1	2
MCCOYS	21/11/2012																									0	0
MCCOYS	21/11/2012		4		5																					2	9
MCCOYS	21/11/2012																1									1	1
MCCOYS	21/11/2012																									0	0
DARCY	21/11/2012																									0	0
DARCY	21/11/2012												1													1	1
DARCY	21/11/2012				1																					1	1
DARCY	21/11/2012		1																							1	1
DARCY	21/11/2012																									0	0
DARCY	5/12/2012																									0	0
DARCY	5/12/2012		21										9										2			3	32
DARCY	5/12/2012		1										3													2	4
DARCY	5/12/2012		2										18													2	20
DARCY	5/12/2012												13													1	13
MCCOYS	5/12/2012		17	3	3							6	28		1				1				1	1		9	61
MCCOYS	5/12/2012		7		1							17	2							1						5	28
MCCOYS	5/12/2012		10	1	13							3	1													5	28
MCCOYS	5/12/2012				1								1													2	2
MCCOYS	5/12/2012		4							1		1									3					4	9

Table 3.3 cont. Macroinvertebrate taxa identified from snag samples from two sites on the lower Goulburn River sampled between October 2012 and February 2013

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SITE	Sample Date	Ceratopogonidae	Chironomidae	Ecnomidae	Elmidae	Empididae	Griptopterygidae	Hydraenidae	Hydrophillidae	Hydroptilidae	Leptophlebiidae	*Micronecta* spp.	Nematoda	Nemertomorpha	Noteridae	Notonectidae	Notonemouridae	*Offadens* spp.	Oligochaeta	*Paratya* sp.	Simuliidae	Staphylinidae	*Tasmanocoenis* spp.	Tipulidae	Veliidae	TOTAL # taxa	TOTAL abundance
DARCY	19/12/2012		43	1	2						1	4												9		6	60
DARCY	19/12/2012		10							1															1	3	12
DARCY	19/12/2012		2										2											1		3	5
DARCY	19/12/2012		9		2														1					1		4	13
DARCY	19/12/2012		2											20												2	22
MCCOYS	19/12/2012		6	2				1					4													4	13
MCCOYS	19/12/2012		162	19	8						7												2			5	198
MCCOYS	19/12/2012		2																							1	2
MCCOYS	19/12/2012	1	3					6										1					1			5	12
MCCOYS	19/12/2012																									0	0
DARCY	16/01/2013		2		1														1							3	4
DARCY	16/01/2013		7							2		1											1			4	11
DARCY	16/01/2013		1		1													1								3	3
DARCY	16/01/2013																									0	0
DARCY	16/01/2013		9									1											1	1		4	12
MCCOYS	16/01/2013		2																1							2	3
MCCOYS	16/01/2013		5		1							2							1			1		2		6	12
MCCOYS	16/01/2013		15		3						3															3	21
MCCOYS	16/01/2013																									0	0
MCCOYS	16/01/2013		45	2	7		1																2			5	57
DARCY	30/01/2013		24	5						1	2													1		5	33
DARCY	30/01/2013		2								1								4							3	7
DARCY	30/01/2013		13	2														1					1			4	17
DARCY	30/01/2013		64	1						1	2												1			5	69
DARCY	30/01/2013		30	2	1					3																4	36
MCCOYS	30/01/2013		64		1				1									1						2		5	69
MCCOYS	30/01/2013		35		1			1				8				1		1	1							7	48
MCCOYS	30/01/2013		26		4							1							1							4	32
MCCOYS	30/01/2013		41	1	1													1								4	44
MCCOYS	30/01/2013		18		1																					2	19
DARCY	13/02/2013		12																							1	12
DARCY	13/02/2013		19						1			1														3	21
DARCY	13/02/2013		9							4		1						5								4	19
DARCY	13/02/2013		2					1				1						1	7				2			6	14
DARCY	13/02/2013		17																4							2	21
MCCOYS	13/02/2013		4								1	1						1								4	7
MCCOYS	13/02/2013		13		1													3								3	17
MCCOYS	13/02/2013		44							3		2						1								4	50
MCCOYS	13/02/2013		44															1			1		1			4	47
MCCOYS	13/02/2013		4		2													1								3	7

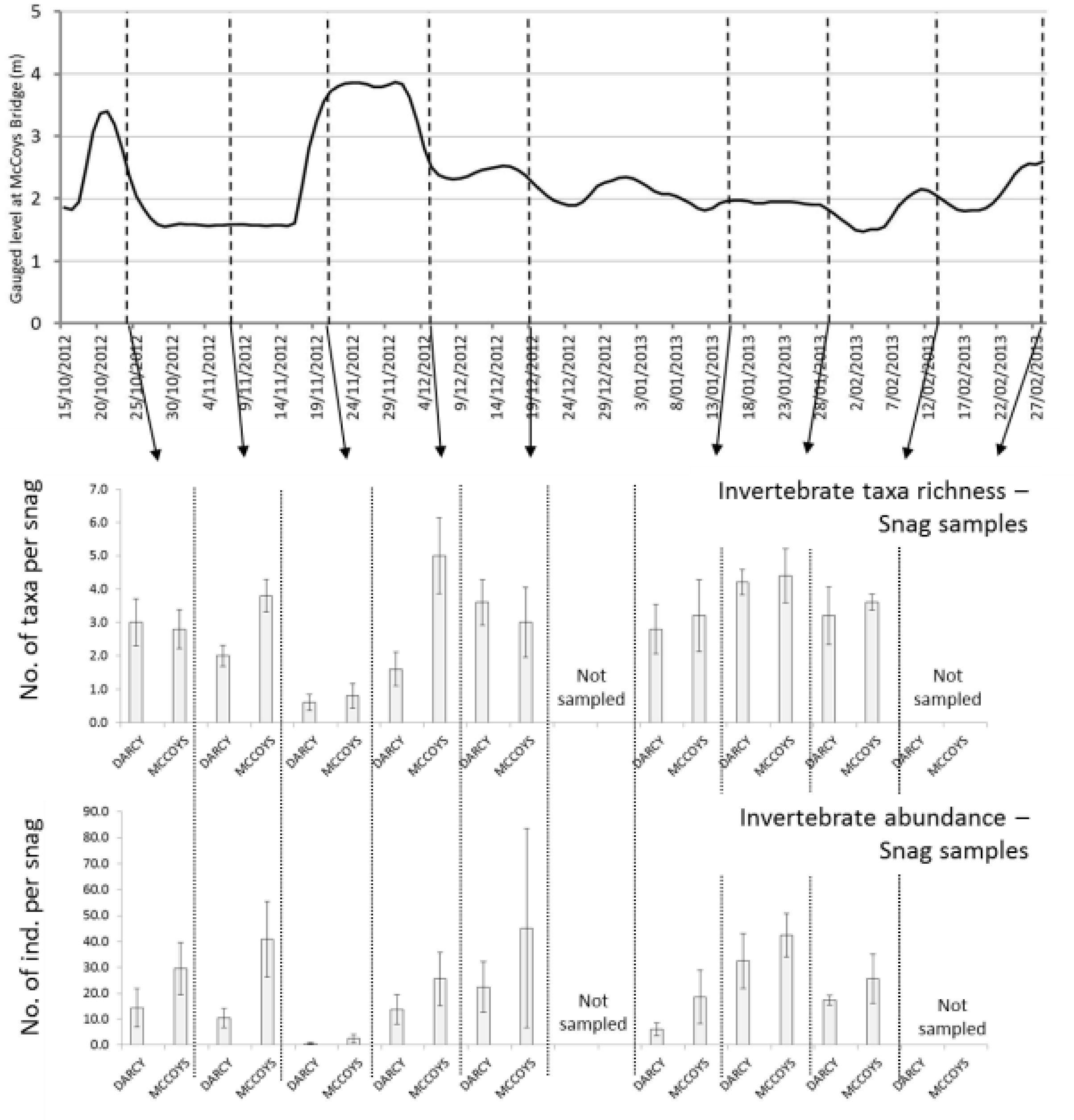
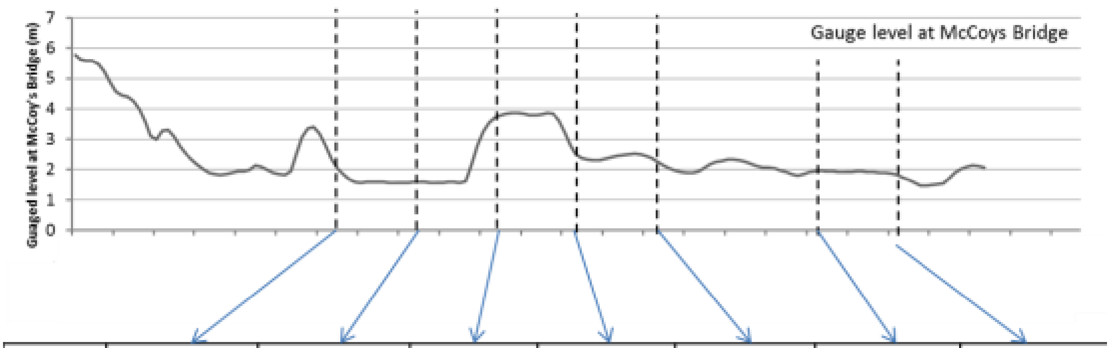


Figure 3.10: Responses of macroinvertebrate taxa richness (middle panel) and abundance (bottom panel) to flow (top panel), measured as means of five snags with standard errors from the lower Goulburn River at two reaches (Darcy and McCoys). Dotted lines and dates indicate dates of sampling



Date Symbol on MDS	24/10/12 ▲	7/11/12 ▼	21/11/12 ■	5/12/12 ◆	19/12/12 ●	16/1/13 ✚	30/1/13 ✖
Charac-teristic taxa	Chironomids Elmidae Leptophlebids	Chironomids Leptophlebids Ecnomidae Empididae	Very low abundance of all taxa	Nematoda Chironomids *Micronecta* Elmidae	Chironomids Tipulidae	Chironomids Elmidae Oligochaetes	Chironomids Ecnomidae Elmidae Leptophlebids

Figure 3.11: Responses of macroinvertebrate abundance in snag samples to flow (top panel), showing taxa identified as characterising the community on each sampling occasion (bottom panel).

Sweep net samples from both reaches were dominated by the corixid bug *Micronecta*, together with chironomids and the shrimp *Paratya* (Table 3.4). Variability between samples within reaches was generally low (greater than 65% similarity in composition within both study reaches). Differences between the two reaches were moderate (45% dissimilarity) with the Darcy reach having higher abundance of the dominant taxa, and lacking the Ecnomidae caddis, which were frequently found at the McCoys reach. Sampling through time revealed no clear relationships between flow and community structure across the sites, with the same taxa predominating through time (Figure 3.12 and Table 3.4). Comparing the samples from the 7/11/2012 (relatively low flows) with those from the 21/11/2012 (relatively high flows) showed that chironomids were at lower abundance during high flows (possibly because of difficulty in sampling) and that *Paratya* abundances were considerably lower. Comparing the 7/11/12 samples with those from 5/12/12 (immediately after a period of high flow) suggested that these results were not simply a consequence of difficulties in sampling at high flows. Chironomids and *Paratya* were both less abundant after the period of high flow and there were slight reductions in abundance of Ecnomidae caddis, *Tasmanocoenis* mayflies. Analysis of similarities found an effect of date of sampling (Global R = 0.707, p=0.001; Figure 3.11) even when differences between sites were taken into account, but there were significant differences between all sampling occasions, regardless of flow conditions. These changes are likely to be due in part to seasonal changes, but it is not possible to determine this absolutely without sampling across multiple years.

Zooplankton

Zooplankton samples were dominated by rotifers and ‘water fleas’ (Ilyocriptidae) numerically, although copepods, ostracods and chydorids were all abundant at some sites on some sampling occasions (Table 3.5). Abundances tended to be higher at the McCoys site than the Darcy site (Figure 3.14). There were no clear patterns of abundance associated with particular flow conditions (Figure 3.14). There was some evidence of a change in community composition after the large environmental flow in November, with rotifers becoming less abundant (Figure 3.14).

Table 3.4: Macroinvertebrate taxa identified from sweep samples from two sites on the lower Goulburn River sampled between October 2012 and February 2013.

SITE	Sample Date	Sample Code	*Micronecta* spp.	Chironomidae	*Paratya* sp.	Ecnomidae	Notonectidae	*Tasmanocoenis* sp.	*Offadens* sp.	Dytiscidae	Griptopterygidae	Tipulidae	Simuliidae	Ceratopogonidae	Oligochaeta	Nematoda	Leptophlebiidae	Leptoceridae	*Hellyethira* sp.	Elmidae	Hydraenidae	Spercheidae	Hydrophillidae	Gerridae	Veliidae	Hydroptilidae	Staphylinidae	Notonemouridae	Total # taxa	Total abundance
DARCY	24/10/2012	DA241012SW1	14	2	2			3																					4	98
DARCY	24/10/2012	DA241012SW2	31	15	12			14	2							2	1												7	164
DARCY	24/10/2012	DA241012SW3	43	18	2			6	10		1		4			3													8	142
MCCOYS	24/10/2012	MC241012SW1	12	14	3				3				22						1										6	121
MCCOYS	24/10/2012	MC241012SW2	1	24	2			1	1				36						1										7	126
MCCOYS	24/10/2012	MC241012SW3	3	28	4				1				24																5	106
DARCY	7/11/2012	DA071112SW1	11	28	7																								3	202
DARCY	7/11/2012	DA071112SW2	123	8	20			1																4					5	271
DARCY	7/11/2012	DA071112SW3	66	31	6			3						1		1								7					7	137
MCCOYS	7/11/2012	MC071112SW1	5	5	4	6										2													5	40
MCCOYS	7/11/2012	MC071112SW2		3	2	10		2	1																				5	45
MCCOYS	7/11/2012	MC071112SW3	7	6	11	1	1							1															6	77
DARCY	21/11/2012	DA211112SW1	44		1				1			1				2				1									6	151
DARCY	21/11/2012	DA211112SW2	94	1	4		1			1																			5	137
DARCY	21/11/2012	DA211112SW3	33									1				2													3	170
MCCOYS	21/11/2012	MC211112SW1	134																										1	141
MCCOYS	21/11/2012	MC211112SW2	5		2																								2	22
MCCOYS	21/11/2012	MC211112SW3	11		3															1									3	312
DARCY	5/12/2012	DA051212SW1	281	10	1		1									3							1						6	577
DARCY	5/12/2012	DA051212SW2	262	2	1		2	1	8			1						2			1								9	903
DARCY	5/12/2012	DA051212SW3	600	7	3							3		1				2						7					7	664
MCCOYS	5/12/2012	MC051212SW1	30		2				8													1							4	298
MCCOYS	5/12/2012	MC051212SW2	226	13	7	2		1	4					1										3					8	465
MCCOYS	5/12/2012	MC051212SW3	193	1	7				4											1				2					6	379
DARCY	19/12/2012	DA191212SW1	72	94	4																			1					4	315
DARCY	19/12/2012	DA191212SW2	100	33	5	1			1						3			1											7	264
DARCY	19/12/2012	DA191212SW3	77	23	9		1							3		3		2		1	1								9	207
MCCOYS	19/12/2012	MC191212SW1	21	20	4 (+	8		1	6				1													30			7	147
MCCOYS	19/12/2012	MC191212SW2	4	6	2	6		2	6				6				3									25			9	97
MCCOYS	19/12/2012	MC191212SW3	14	5	5			1	6								1			1						4			8	229
DARCY	16/01/2013	DA160113SW1	163	20	4			1	1						2						1								7	422
DARCY	16/01/2013	DA160113SW2	161	67	2																								3	337
DARCY	16/01/2013	DA160113SW3	88	11	3										1								1	3					6	167
MCCOYS	16/01/2013	MC160113SW1	1	5	2			4	30															3		13	2		8	123
MCCOYS	16/01/2013	MC160113SW2	16	11	5			2	19			1			1		3									4	1		10	156
MCCOYS	16/01/2013	MC160113SW3	8	33	6			3	20								3	2								18			8	491
DARCY	30/01/2013	DA300113SW1	257	104	5	4		19	1						1		3	2						1		1			11	662
DARCY	30/01/2013	DA300113SW2	150	81	13	3		7	4			1					1	2		1								1	11	537
DARCY	30/01/2013	DA300113SW3	95	109	2	6		24	32									1								4			8	433
MCCOYS	30/01/2013	MC300113SW1	11	61	5			14	56				2				10								1				8	195
MCCOYS	30/01/2013	MC300113SW2	1	12	1	1			13							1				1	1					4			9	150
MCCOYS	30/01/2013	MC300113SW3	5	38	7	1		9	27				8				1			2						17			10	329
DARCY	13/02/2013	DA130213SW1	64	127	6			8	6						3														6	395
DARCY	13/02/2013	DA130213SW2	119	53				1	3						2											3			6	358
DARCY	13/02/2013	DA130213SW3	113	54	4				1						3											2			6	231
MCCOYS	13/02/2013	MC130213SW1	8	21	2				15						1									1		6			7	111
MCCOYS	13/02/2013	MC130213SW2	1	7					21				9							1						18			6	133
MCCOYS	13/02/2013	MC130213SW3	13	12	12			1	8																	30			6	76

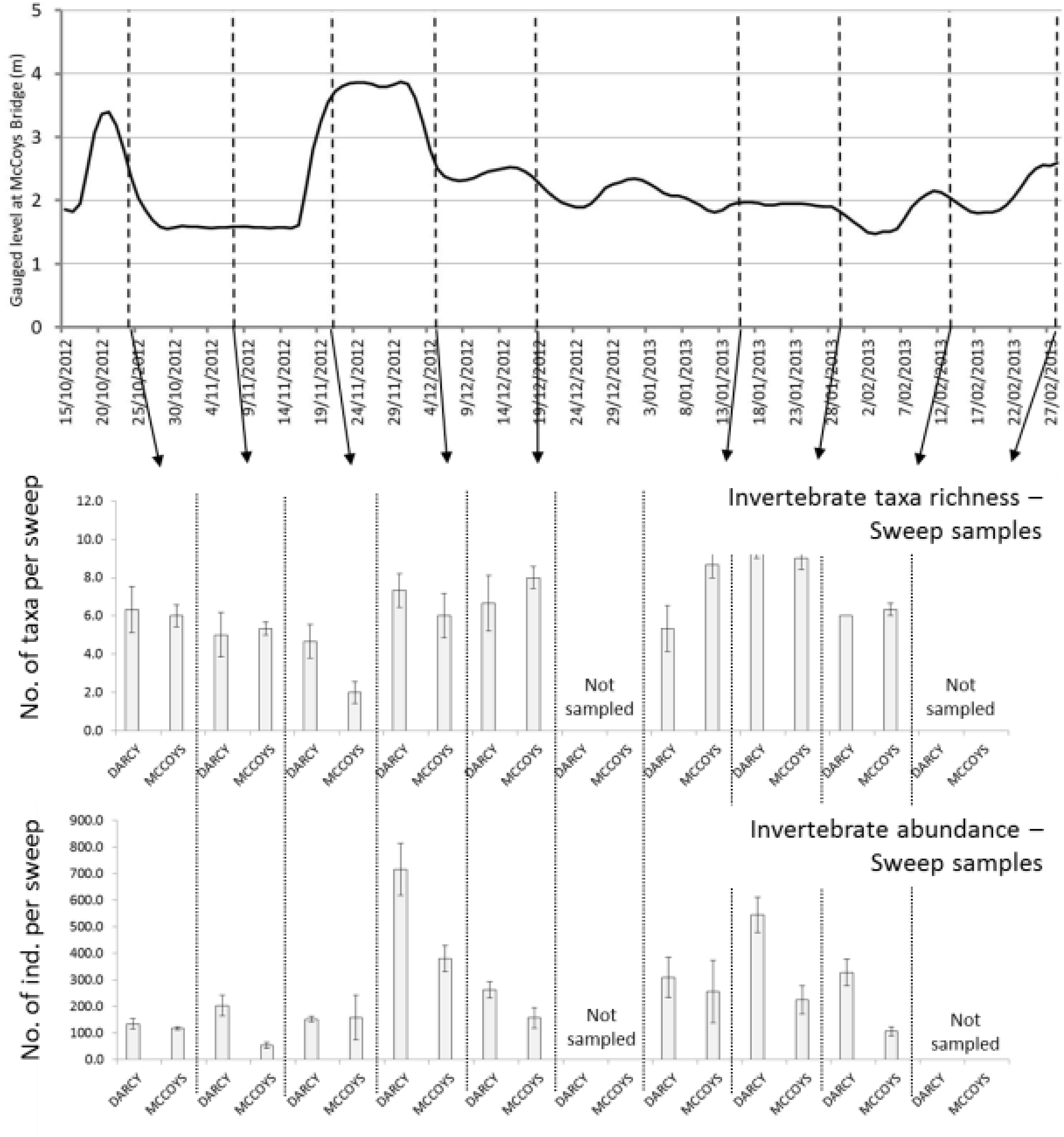
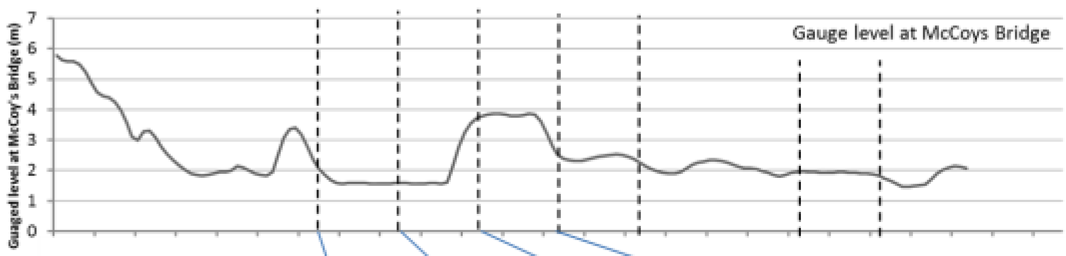


Figure 3.12: Responses of macroinvertebrate taxa richness (middle panel) and abundance (bottom panel) to flow (top panel), measured as means of three sweep samples with standard errors from the lower Goulburn River at two reaches (Darcy and McCoys). Dotted lines and dates indicate dates of sampling.



Sampling date Symbol on MDS	24/10/12 ▲	7/11/12 ▼	21/11/12 ■	5/12/12 ◆
Characteristic taxa	Chironomidae *Micronecta* spp. Simuliidae *Paratya* sp. Tasmanocoenis spp.	Chironomidae *Paratya* sp. *Micronecta* spp. Ecnomidae	*Micronecta* spp. *Paratya* sp.	*Micronecta* spp. *Paratya* sp. *Offadens* spp. Chironomidae

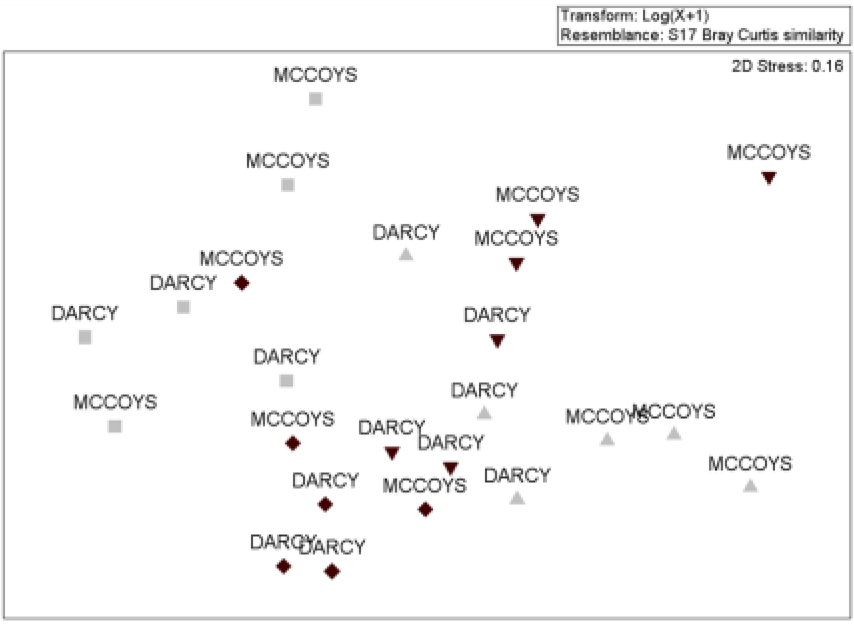


Figure 3.13 Responses of macroinvertebrate abundance in sweep net samples to flow (top panel), showing taxa identified as characterising the community on each sampling occasion (middle panel), and a multi-dimensional scaling ordination of similarities in communities between sites and sampling dates (bottom panel).

Table 3.5: Zooplankton taxa identified from zooplankton hauls from two sites on the lower Goulburn River sampled between October 2012 and February 2013.

SITE	DATE	SITE CODE	Rotifera	Ilyocriptidae	Copepoda	Ostracoda	Chydoridae	Oribatidae	TOTAL abundance
MCCOYS	24/10/2012	MC24102012ZOO1	46	5	4	2	0	0	57
MCCOYS	24/10/2012	MC24102012ZOO2	6	11	2	0	0	1	20
MCCOYS	24/10/2012	MC24102012ZOO3	28	13	7	2	0	1	51
DARCY	24/10/2012	DA24102012ZOO1	112	6	8	1	0	0	127
DARCY	24/10/2012	DA24102012ZOO2	26	13	7	2	0	1	49
DARCY	24/10/2012	DA24102012ZOO3	31	7	2	1	0	0	41
MCCOYS	7/11/2012	MC07112012ZOO1	300	30	6	1	0	0	337
MCCOYS	7/11/2012	MC07112012ZOO2	108	20	2	0	0	0	130
MCCOYS	7/11/2012	MC07112012ZOO3	44	12	1	0	0	0	57
DARCY	7/11/2012	DA07112012ZOO1	33	12	1	1	0	0	47
DARCY	7/11/2012	DA07112012ZOO2	6	4	0	2	9	0	21
DARCY	7/11/2012	DA07112012ZOO3	22	8	0	3	18	0	51
MCCOYS	21/11/2012	MC21112012ZOO1	0	0	23	1	2	0	26
MCCOYS	21/11/2012	MC21112012ZOO2	0	2	15	0	8	0	25
MCCOYS	21/11/2012	MC21112012ZOO3	0	0	62	0	0	0	62
DARCY	21/11/2012	DA21112012ZOO1	9	6	2	12	0	0	29
DARCY	21/11/2012	DA21112012ZOO2	4	12	0	12	0	0	28
DARCY	21/11/2012	DA21112012ZOO3	4	8	1	43	0	2	58
MCCOYS	5/12/2012	MC05122012ZOO1	4	13	8	0	26	0	51
MCCOYS	5/12/2012	MC05122012ZOO2	0	16	1	1	0	0	18
MCCOYS	5/12/2012	MC05122012ZOO3	2	48	4	2	0	1	57
DARCY	5/12/2012	DA05122012ZOO1	0	34	6	4	1	0	45
DARCY	5/12/2012	DA05122012ZOO2	1	21	3	3	0	0	28
DARCY	5/12/2012	DA05122012ZOO3	0	3	2	0	0	0	5
MCCOYS	19/12/2012	MC19122012ZOO1	71	49	9	2	6	0	137
MCCOYS	19/12/2012	MC19122012ZOO2	95	53	3	3	4	0	158
MCCOYS	19/12/2012	MC19122012ZOO3	94	46	8	1	0	0	149
DARCY	19/12/2012	DA19122012ZOO1	6	15	0	6	0	0	27
DARCY	19/12/2012	DA19122012ZOO2	16	0	1	3	0	0	20
DARCY	19/12/2012	DA19122012ZOO3	8	25	5	13	0	0	51
MCCOYS	16/01/2013	MC16012013ZOO1	106	125	10	3	0	1	245
MCCOYS	16/01/2013	MC16012013ZOO2	268	229	13	0	0	1	511
MCCOYS	16/01/2013	MC16012013ZOO3	112	150	10	1	0	0	273
DARCY	16/01/2013	DA16012013ZOO1	1	28	2	0	0	0	31
DARCY	16/01/2013	DA16012013ZOO2	14	80	7	1	0	0	102
DARCY	16/01/2013	DA16012013ZOO3	6	53	3	2	0	0	64
MCCOYS	30/01/2013	MC30012013ZOO1	390	16	10	1	115	1	533
MCCOYS	30/01/2013	MC30012013ZOO2	254	4	6	0	53	0	317
DARCY	30/01/2013	DA30012013ZOO1	6	7	1	1	26	1	42
DARCY	30/01/2013	DA30012013ZOO2	3	4	2	0	5	0	14
MCCOYS	13/02/2013	MC13022013ZOO1	99	84	8	0	15	1	207
MCCOYS	13/02/2013	MC13022013ZOO2	54	42	3	0	0	0	99
MCCOYS	13/02/2013	MC13022013ZOO3	92	116	12	0	2	0	222
DARCY	13/02/2013	DA13022013ZOO1	6	10	3	0	7	0	26
DARCY	13/02/2013	DA13022013ZOO2	9	12	3	0	19	0	43
DARCY	13/02/2013	DA13022013ZOO3	3	8	0	3	2	0	16

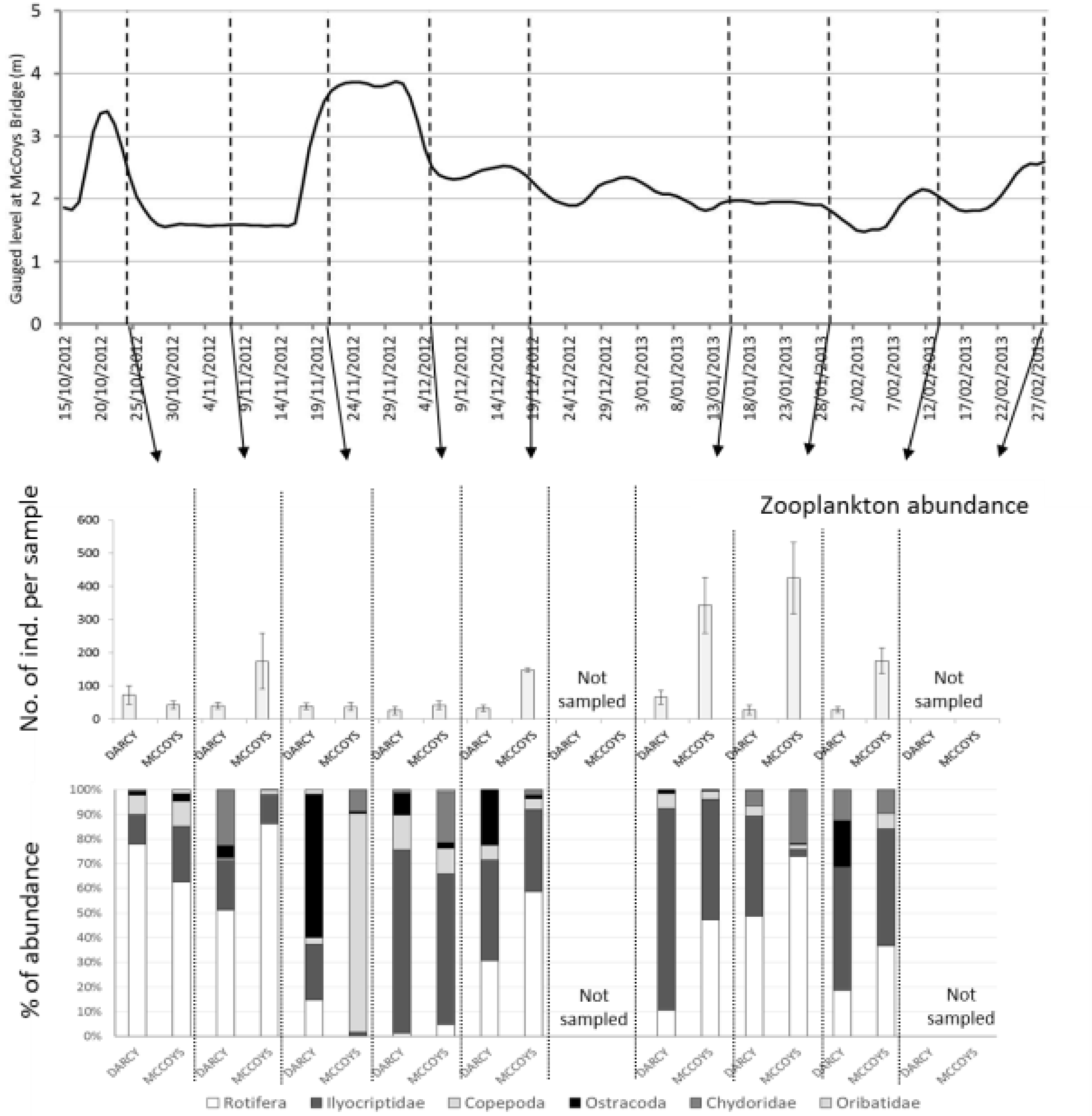


Figure 3.14 Responses of zooplankton abundance (middle panel) and community composition (bottom panel) to flow (top panel), measured as means of three haul samples with standard errors from the lower Goulburn River at two reaches (Darcy and McCoys). Dotted lines and dates indicate dates of sampling. Taxa are shown in order from the bottom to the top of the bar as indicated in the legend.

3.4 Discussion

Planktonic chlorophyll-a concentrations over the study period were low compared to levels reported for other lowland rivers (20–200 mg/m^3; Reynolds & Descy 1996), including the very similar Edward-Wakool system in New South Wales (Watts et al. 2013). Primary production within the water column is unlikely to be the main source of carbon in these systems, but during lower flows in October gross primary production did exceed ecosystem respiration. The benthic chlorophyll-a levels in these systems did not exceed the 100 mg chlorophyll-a /m^2 level recommended by Quinn (1991) for nuisance biofilm and are similar to those observed in similar rivers in south eastern Australia (Watts et al. 2005; Watts et al. 2008). Biofilm composition was also generally consistent with both those studies and the more recent work of Watts et al. (2013). There was some evidence for an effect of high flow events on productivity of the lower Goulburn channel. Values for both gross primary productivity (GPP) and ecosystem respiration (ER) were typical of those found in large slow flowing rivers world-wide, and are strikingly similar to the values recorded from the physically similar Edward-Wakool system in NSW (Watts et al. 2013). The large environmental flow in November was associated with a shift from in-channel (photosynthetic) productivity to energy derived from outside the channel. This was due in part to the environmental flows in late November 2012 reducing benthic algal biomass, particularly in deeper water. This is likely to be due to shading rather than physical removal, as chlorophyll-a values (indicating actively photosynthesising tissue) declined more than values for total biomass. Analysis of light and turbidity data suggests that light limitation is likely to be an important factor underlying these changes (see Section 3.2, this report). There was no evidence of any effect on water-borne primary producers (phytoplankton), which were present at detectable levels, but did not alter consistently with changes in flow. The findings are also consistent with several investigations of the effects of discharge variability on biofilm biomass and productivity (Watts et al. 2005, 2008).

Zooplankton communities were dominated by rotifers, which is a typical feature of lowland rivers in the Murray-Darling Basin (Shiel et al. 1982; Ning et al. 2012; Watts et al. 2013). Small-bodied zooplankton are favoured in these systems because of their short generation times (Shiel et al. 1982). In the current study there was some evidence of flow effects on zooplankton communities in terms of community composition but not abundance, with rotifer densities lower during the large environmental flow in November. These effects can be due to dilution effects from increasing volumes of water (Basu and Pick 1997) but the dramatic change in rotifer numbers relative to other animals suggests they are responding to high flows, either from being advected from upstream or due to a rapid reproduction response.

Changes in rotifer numbers were not observed in a study of a similar system in NSW (Watts et al. 2013), and it may be that some other factor drove the pattern observed here. The relatively simple nature of the channel in both study reaches, with relatively few slackwater areas, may not favour zooplankton (Watts et al. 2013).

Macroinvertebrate communities were similar to those observed in other lowland rivers in the Murray-Darling Basin, and were dominated by generalist taxa both in terms of vulnerability to flows and dietary habits. Previous studies have suggested that reductions in flow variability due to river regulation or drought can reduce macroinvertebrate biodiversity (Pardo et al. 1998; Poff et al. 1997; Bunn and Arthington 2002). There was no clear effect of environmental flows on macroinvertebrate density or diversity in this study, however the system has been affected by a number of large flows over the last two years, and it may be that the fauna has already responded to those events. There was evidence of a reduction in macroinvertebrate biomass after high flows in November 2012. Measurements during the high flows may not be reliable, as it was difficult and dangerous to access suitable habitat for sampling. However macroinvertebrate samples taken when the flows had dropped suggested that the high flow events had reduced macroinvertebrate biomass, particularly for a number of taxa that are relatively weak swimmers. The community appeared to recover quickly (weeks) and to a composition that was similar to that prior to the high flows.

It appears that a large environmental flow in early summer was associated with a change in the basal productivity of the food web, and had effects on both the abundance and diversity of macroinvertebrate communities associated with snag habitat. However smaller environmental flows in late summer did not appear to have detectable effect on either basal productivity or macroinvertebrates. It is not possible to determine whether this is due to the timing of the flow or the magnitude. However it seems likely that small environmental flows in the relatively simple U-shaped channel of the lower Goulburn are largely associated with a slight deepening of water, with relatively minor impacts on algal communities and on organic matter supply. The larger flow in November may have inundated a number of in-channel steps and introduced more organic matter into the flow, resulting in increased ecosystem respiration and flow-on impacts for macroinvertebrate communities. Algal communities in contrast are likely to have been affected by sloughing of biofilms due to increased flow, and shading as a result of deeper water, thereby reducing gross primary production. It is also possible that higher flows resuspend algal cells from the stream bed, increasing organic matter in the channel and generating a lagged response in algal productivity.

4 MOVEMENT OF ORGANIC MATTER

4.1 Evaluation question and expected outcomes

This section addresses the evaluation question: What has been the effect of environmental flows on release of organic matter? The expected outcome was that environmental flows delivered as freshes would mobilise organic matter increasing availability as a basal resource for the river ecosystem.

4.2 Methods

Organic matter supply and retention

<u>Movement of marked leaves</u>

To assess mobilisation and entrainment of leaves into the channel during environmental flows, marked leaf strips were deployed and surveyed at biweekly intervals (Figure 4.1). Strips were placed at three locations at each site (0m, 25m and 50m, see Figure 4.2). Red gum leaves were painted with fluorescent paint of different colours. These are laid as a continuous 20 cm wide strip extending 2 m from the water's edge. Leaves were placed in alternating 25 cm bands of different colours. Each leaf strip was photographed initially and then biweekly, before being replaced. Leaves were recorded as 'unmoved', 'moved' (not in their original location but still in the shot (within 1 m of original location) or 'lost' (not visible in the shot). The percentage of leaves in each class was recorded.

<u>Drift nets.</u> To assess organic matter movement during environmental flow events three drift nets were set out at the top of each study reach for one hour on each of the biweekly sampling occasions. Drift nets had a circular opening of 0.2 m diameter and a mesh size of 250 µm. Proportion of cross section of river sampled <0.001%. Material was retrieved from the nets, dried and weighed to provide an estimate of the amount of organic matter that is mobilised under different flow conditions. These data are represented as the total amount of material caught in each net per hour the nets were deployed.

<u>Marking and relocation of snags.</u>

At both study sites a set of snags were permanently marked using aluminum plant tags. For both sites ten snags were chosen within three classes; small (length <1 m, diameter of largest piece 5-10 cm), medium (1-2 m, diameter of largest piece 15-20 cm) and large (2-5 m, diameter of largest piece >20 cm). After each flow event the location of each snag was determined and changes recorded. Snags were classified as 'unmoved', 'moved' (within 10 m of original location) or 'lost (not relocatable).

Turbidity

A turbidity meter was used to measure the amount of light penetrating the water biweekly at each reach. Within each reach measurements were taken at three locations. At each location readings were taken at 10, 20, 50 and 100 cm depth. The three values for each depth at each reach on each date were averaged.

Light

Continuous light measurements were taken at 5 cm and 50 cm below baseflow, and at 50 cm above baseflow. These were equivalent to gauge levels at McCoys Bridge of 2 m (shallow), 1.5 m (mid) and 1 m (deep).

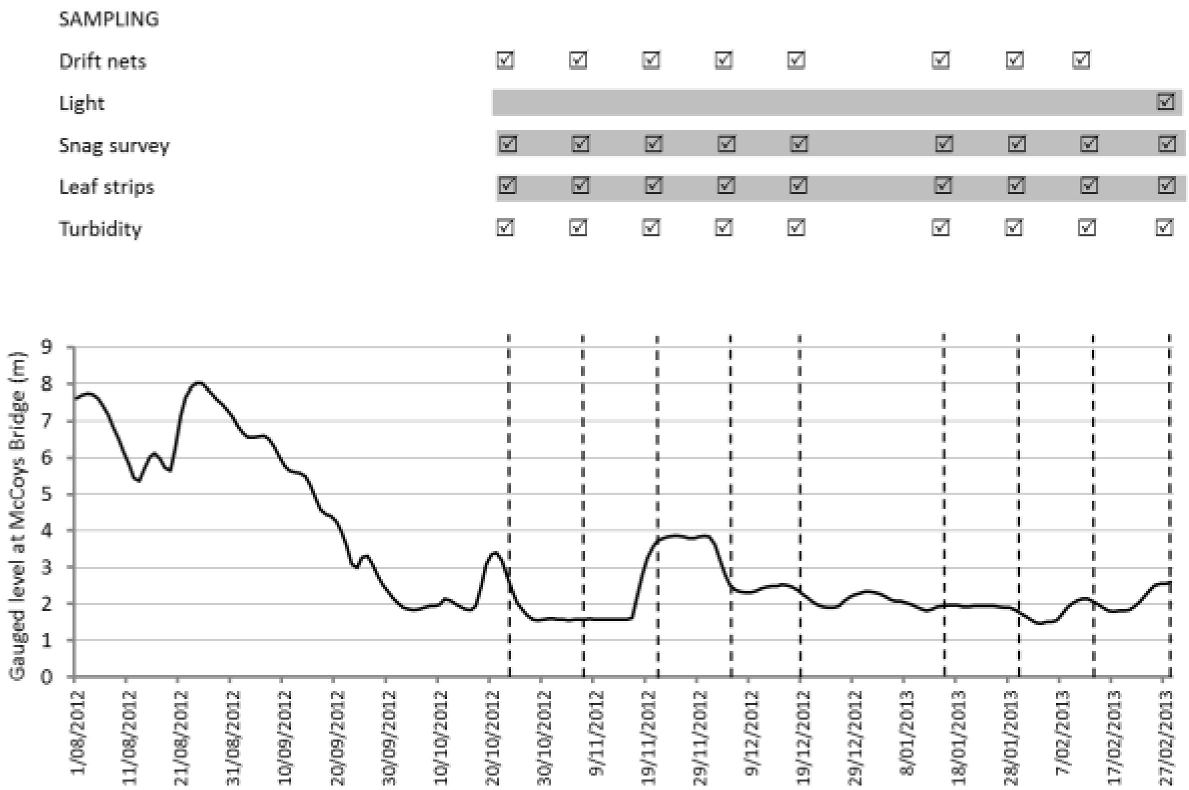


Figure 4.1: Timing of sampling (dotted vertical lines) for measures of organic matter retention at Darcy Track and McCoys Bridge on the lower Goulburn River. Grey shading indicates continuous measurement. Boxed ticks indicate sampling which is reported on in this report.

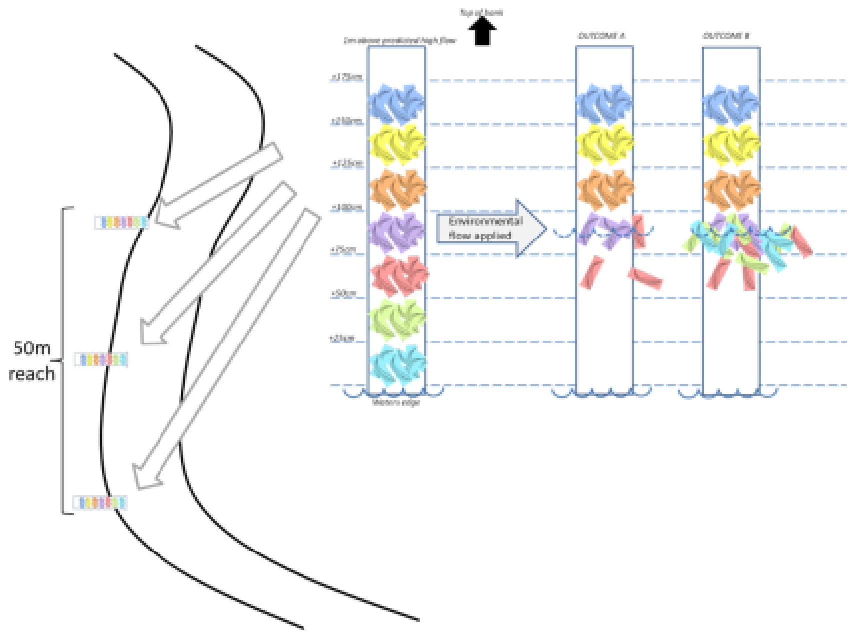


Figure 4.2: Experimental layout of leaf mobilisation strips. Strips provide an index of organic matter mobilisation due to environmental flows. In Outcome A leaves have been mobilised into the channel and are available for aquatic organisms. In Outcome B leaves have been moved but have been deposited back into the terrestrial environment as flows recede.

4.3 Results

Organic matter supply and retention

<u>Movement of marked leaves.</u>

The marked leaf strips were deployed to determine when flow energy was sufficient to move leaves into the channel, and to determine what percentage of leaves were moved back into the channel as opposed to settling out near (within 1 m) of their original location. The method proved generally reliable, with the largest flows removing the strips entirely (compare Figure 4.3 A and B), baseflows leaving the strips largely intact (Figure 4.3 C and D), and smaller flows leaving some leaves intact (Figure 4.3 E and F). The late November 2012 sampling was not possible because of high flows. The large environmental flow in November removed all of the leaf litter from each strip and no marked leaves were visible along the shore (Figure 4.4). The smaller environmental flows in early 2013 did leave a small proportion of the leaves in place, and it was evident that some component of the leaf litter was moved and then settled out again, likely as flows dropped. These smaller flows left ‘drift lines’ of marked leaves along the study reach.

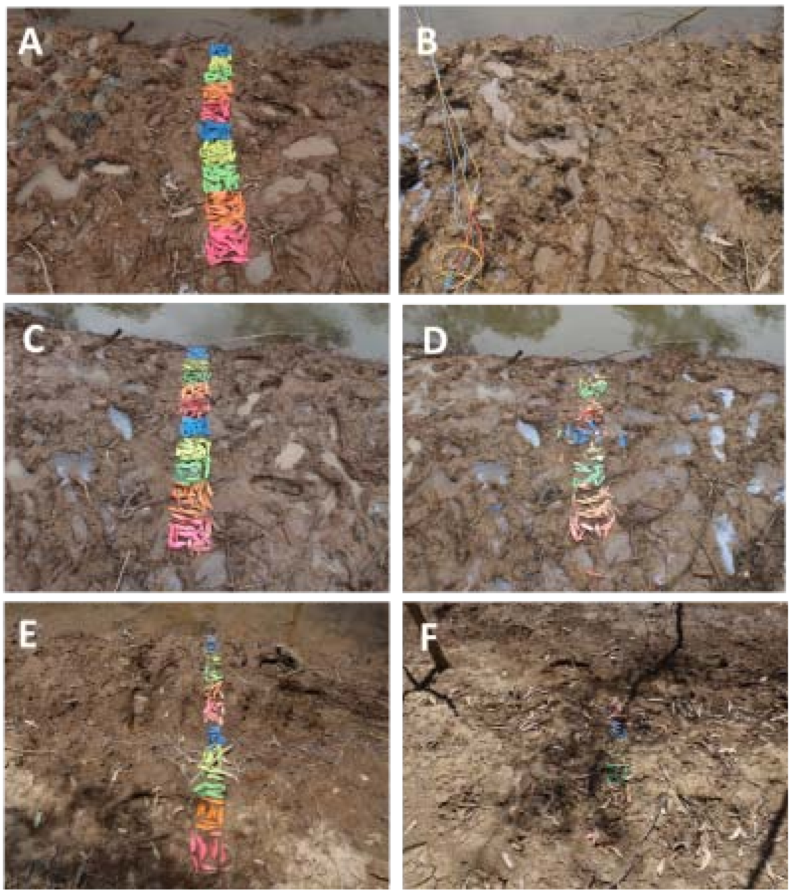


Figure 4.3: Examples of leaf strips responses to flows at Darcy Track. A and B; Darcy Track 45 m strip before (7/11/12) and after (5/12/12) a large environmental flow. C and D; Darcy Track 45 m strip before (16/1/13) and after (30/1/13) a period of stable flows. E and F; McCoys 15 m strip before (30/1/13) and after (30/2/13) a small environmental flow.

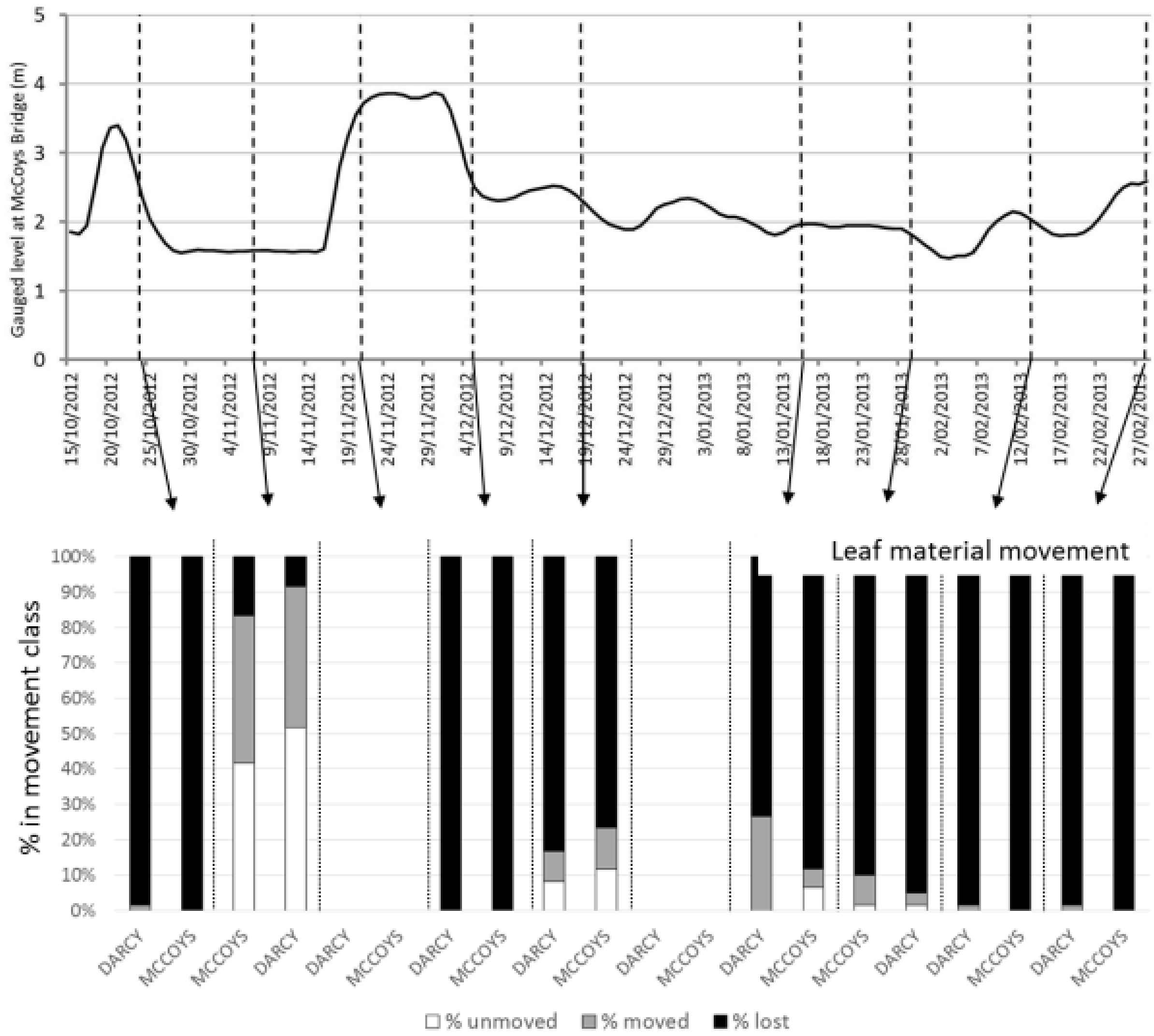


Figure 4.4: Movement of leaves in indicator strips (bottom panel) in response to flows (top panel) at two sites on the lower Goulburn River, McCoys and Darcy.

<u>Drift nets.</u>

Amounts of organic and inorganic material in the water column did appear to vary with flow. There was a peak in material in the flow at the McCoys reach during the high flows of late November 2012, although this appeared short lived (Figure 4.5). It is interesting how much less organic material is contained in the coarser particulate fraction. It is possible that the timing of the placement of the nets changed the amount of material captured, if coarse particulate organic material was mobilised shortly after the flows increased and was then depleted by settlement until another change in flow.

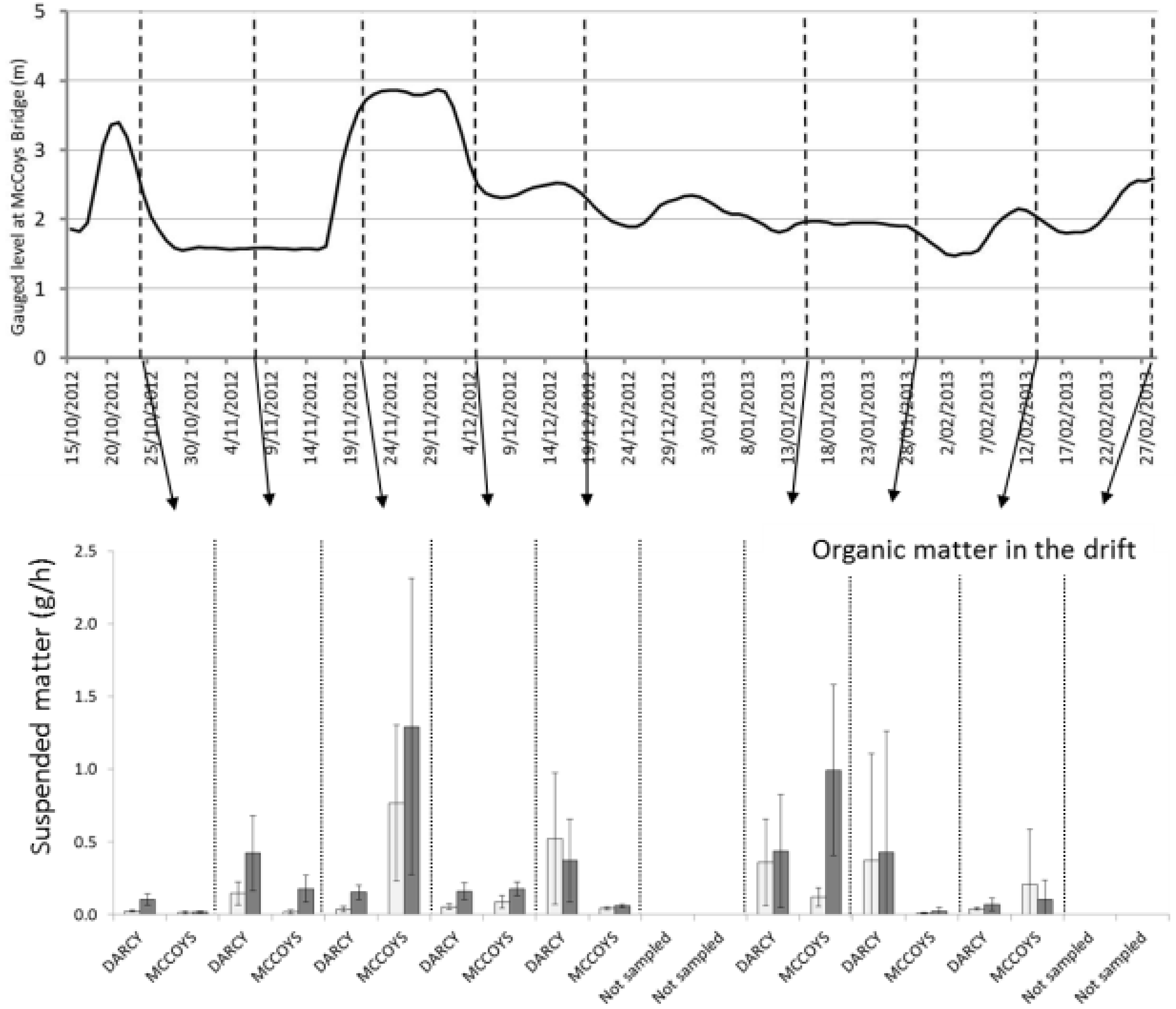


Figure 4.5: Patterns in suspended material in the flow captured using drift nets collected from the lower Goulburn River at two reaches (Darcy and McCoys) as flow varies (top panel). Dotted lines and dates indicate dates of sampling. Dark grey bars indicate inorganic material and pale bars indicate organic material.

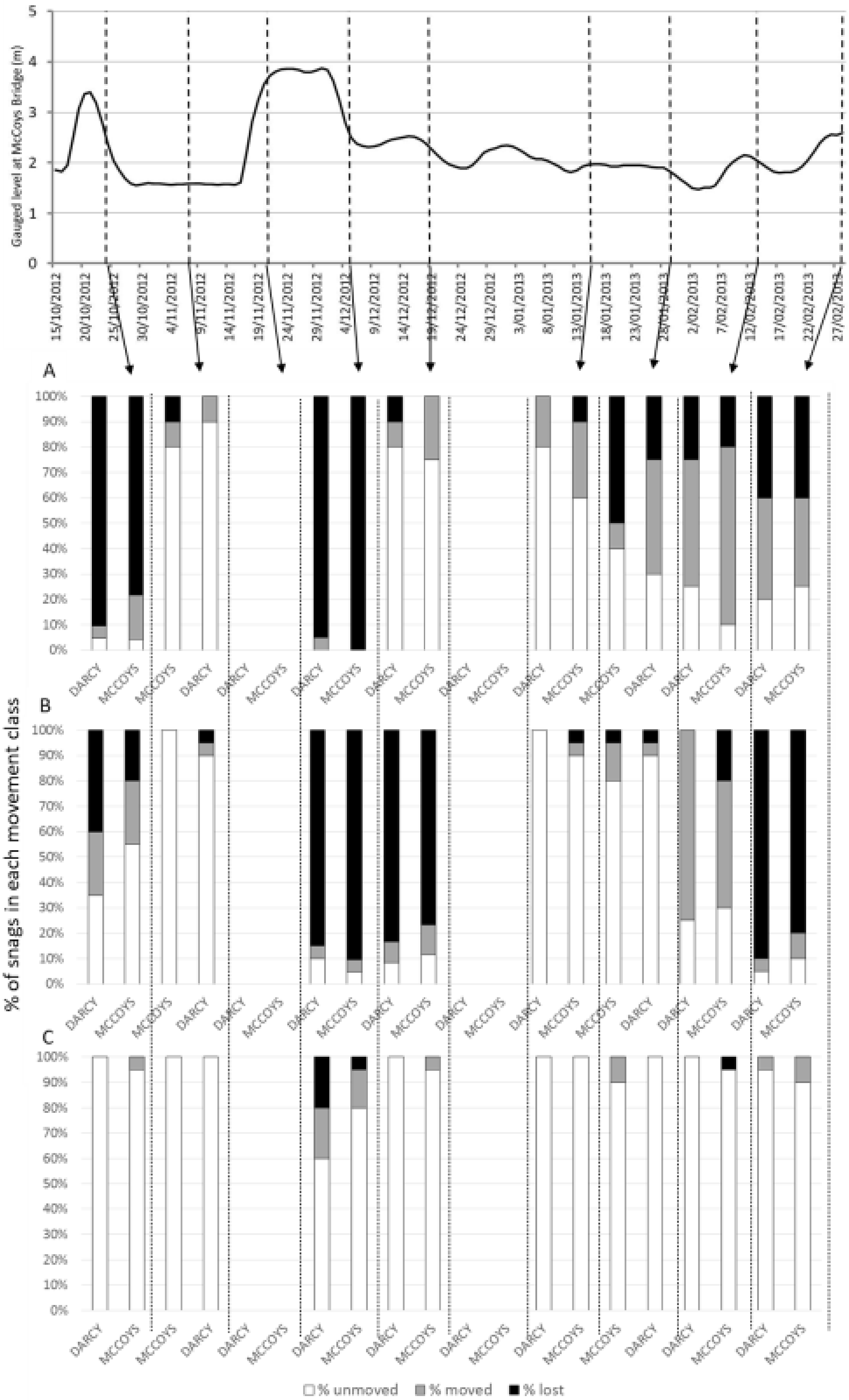


Figure 4.6: Percentage of snags of different sizes that moved with different flows (top panel) in the lower Goulburn River at two sites; Darcy and McCoys. A) snags <1 m long, B) 1-2 m long, C) 2-5 m long.

Marking and relocation of snags.

Results for the snag data were highly consistent with that from the leaf strip data. All of the environmental flows delivered over the study period were capable of moving small snags, although there was also some evidence that small pieces of timber were being moved by human activities (largely for use as fishing poles). The large environmental flow in November moved almost all of the small pieces of wood, a large proportion of the medium sized pieces, and some of the large pieces (Figure 4.6). The smaller flows in early 2013 were capable of moving small pieces of wood (<1 m long) but generally did not move larger pieces. This confirms that environmental flows can move woody debris of the size found in the channel of the lower Goulburn River, but that larger flows are needed to shift the large more stable pieces of timber.

Turbidity

There were clear differences in turbidity between the sites, with McCoys having much more turbid water (Figure 4.7). This study was not able to, nor sought to, identify the source of the differences between sites, but was rather interested in the variability within sites and any correlations with river levels. Patterns with depth were not strong, suggesting that the water column is relatively well mixed. There is some evidence of an effect of flow, with slightly more turbid water as flows increased in late November 2012, however this effect seems to persist beyond the freshes (perhaps because of resuspension of bed materials) and does not occur for smaller flows.

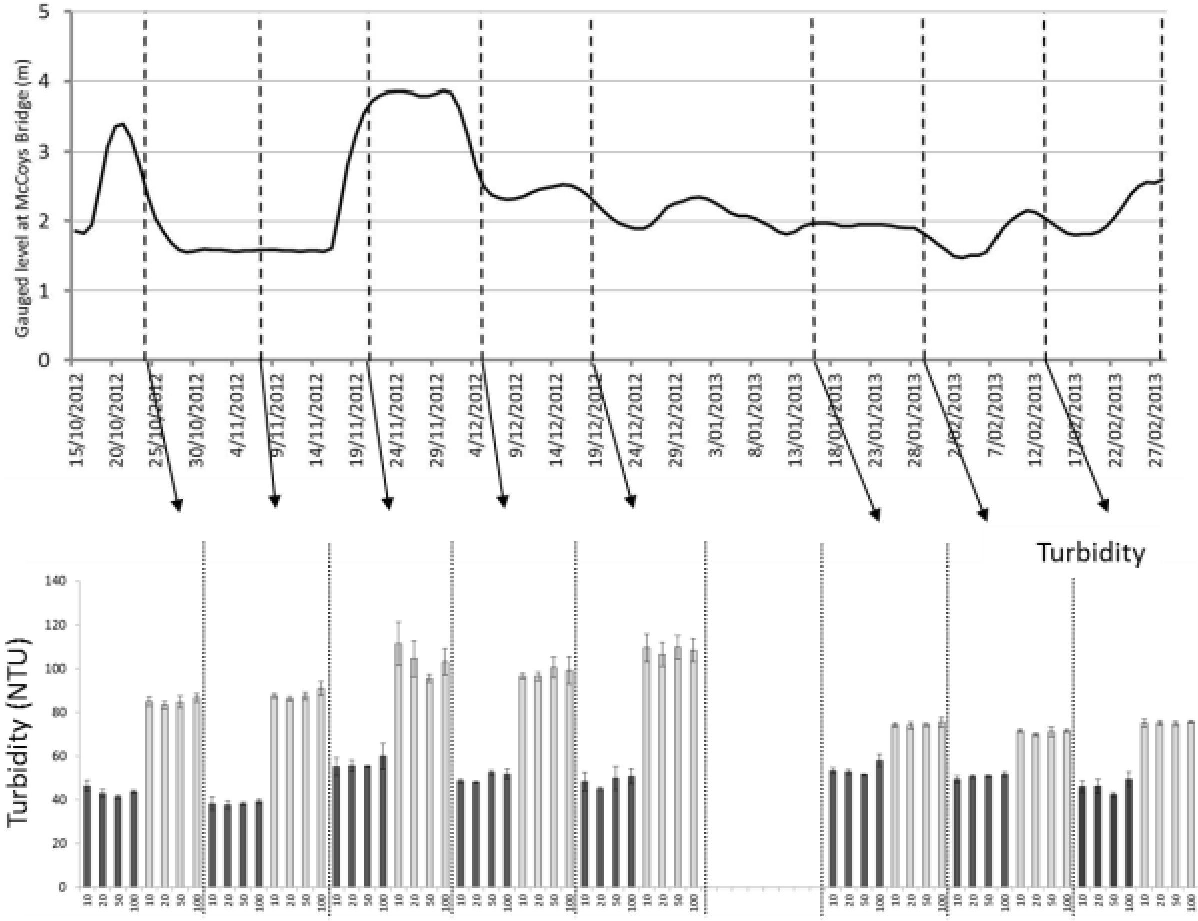


Figure 4.7: Patterns in water turbidity at different depths from the lower Goulburn River at two reaches (Darcy and McCoys) as flow varies (top panel). Dotted lines and dates indicate dates of sampling. Dark grey bars indicate measurements from the Darcy reach, while light grey indicates measurements from the McCoys reach. Values are averages of three measurements from each depth with standard errors.

Light

Analysis from the light loggers at different depths clearly revealed that there was rapid attenuation of light in the first metre of water depth. As a result, when water depths increased due to environmental flows (particularly in November 2012) very low levels of light reached the stream bed where the algal colonisation trays were placed (Figure 4.8). Early in the study period, the 'shallow' light sensor was exposed, meaning that values were very similar to those for the 'mid' sensor which was at the water surface. As the water deepened, all sensors were covered. Attenuation of light occurred quickly, with intensity at 50 cm depth four times weaker than at surface (median change in light from shallow to mid sensor).

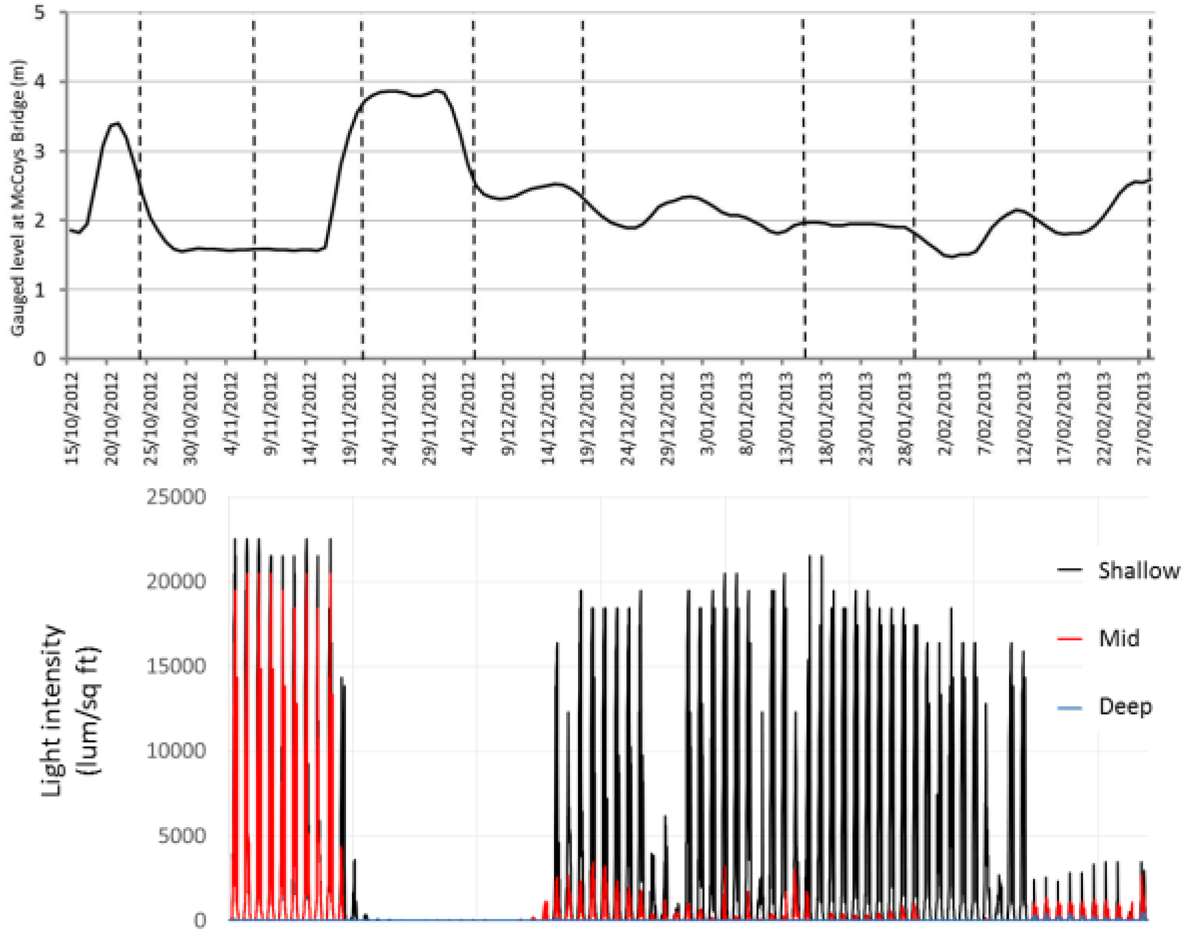


Figure 4.8: Light intensity reaching the stream bed at three fixed locations as water levels vary in the lower Goulburn River. Shallow indicates a sensor at a 2 m gauge height on the McCoys Bridge gauge, mid is 1.5 m and deep 1 m. The time axis for the lower panel is aligned with that on the upper panel.

4.4 Discussion

The environmental flow delivered in November 2012 was sufficient to mobilise large amounts of organic matter from the banks, including wood and snags more than 2 m in length. This is likely to have increased the amount of organic matter in the channel available for decomposers. Changes in organic matter loads to the channel is likely to have explained some of the variability seen in ecosystem respiration described in Section 3.1 and are likely to be an important driver of downstream processes. Deepening water over biofilms will have resulted in light limitation, reducing gross primary production. This may be a particularly important impact of environmental flows – particularly when flows are of short duration. The scenario may lead to the shading of benthic algal communities already present, but not allow sufficient time for algae to colonise newly inundated habitat along the margins. This may result in lower quality of algae as a food resource for grazers (due to light limitation), and could potentially lead to food limitation for some grazers. The response time for primary productivity on newly wetted river margins is not well known and warrants further study, particularly with respect to the effects of fluctuating water levels.

While algal communities are important, the data shown in Section 3.1, and in other published studies (Watts et al. 2013) shows that there is a strong reliance on terrestrial organic matter as a basal resource for stream productivity. Woody debris and leaves are important substrates for algal growth, and as a resource in themselves (Watts et al. 2005). Mobilisation of leaves and wood into the channel was achieved by environmental flows in this study, with surprisingly large material mobilised by the flows in November 2012.

Environmental flows affect organic matter dynamics in the lower Goulburn River through mobilising wood and leaf material. If this material is retained in the wetted channel this will provide food resources and habitat for animal taxa. However changes in light regimes may also contribute to a reduction in algal resources in channel.

5 BANK VEGETATION

5.1 Evaluation question and expected outcomes

This section addresses the evaluation question: Have environmental flow freshes promoted flood tolerant vegetation on the stream bank in the lower Goulburn River? The expected outcome was that environmental flows delivered as freshes would favour flood tolerant vegetation normally found on a river bank and reduce the cover of terrestrial vegetation which has been able to encroach into the channel as a result of extended low flows periods.

5.2 Introduction

The effects of flow regulation on riparian vegetation assemblages have been well documented around the world (Nilsson and Svedmark, 2002). Specific effects of flow regulation include a shift in plant species composition towards more opportunistic 'weedy' taxa that are able to cope with an altered flow regime, compared to endemic species adapted to a river's natural flow regime. Opportunistic taxa are often exotic, and so regulated rivers are characterised by a higher proportion of exotic species in their riparian zones (Greet et al., 2013).

There are fewer (if any) studies of the response of riparian vegetation to flow restoration – i.e. the re-instatement of a more natural flow regime through environmental flows. One reason for this might be that vegetation assemblages are an 'integrative' response to environmental watering; vegetation assemblages change relatively slowly, and reflect the history of inundation and watering regimes rather than specific events. As such, it would be difficult to directly measure (in the field) responses to individual short-term releases of Commonwealth environmental water. Rather, improved vegetation assemblages will result from an improved flow regime that may include many sources of water. With the above considerations in mind, here we attempt to infer the effects for riparian vegetation assemblages of a more variable flow regime – typical of what is possible with environmental flow regimes – compared to an invariant flow regime that might otherwise occur under highly flow-regulated conditions. More specifically, we compare the stream bank vegetation of the lower Goulburn River, which receives environmental flow allocations from Commonwealth Environmental Water holdings, to that of the lower Broken Creek, which experiences almost constant water elevations because of its sequential weir pool structure. We also employ 3 years of vegetation monitoring data from the Goulburn river to quantify models of vegetation response to different inundation patterns, and use these models to estimate the effect of different flow regimes on vegetation cover, including what we would expect to see had no environmental flows been delivered in the Goulburn during 2012.

5.3 Conceptual Model

We hypothesise that improved baseflows, and the existence of freshes in the lower Goulburn River will lead to differences in riverbank vegetation assemblages compared to what would have occurred otherwise. To test this hypothesis, we will compare vegetation assemblages on the banks of the lower Goulburn River to those within weir pools on the lower Broken Creek. The sequential weir pool structure of Broken Creek provides near-constant water levels, which is a reasonable analogue of what would occur during the growing season (spring – early summer) in the lower Goulburn River if it were not for environmental flow releases. We will also be able to compare vegetation assemblages in 2012 to those sampled on the lower Goulburn River in 2008 and 2010. 2008 was much drier than 2010 and 2012, providing a further indication of the effect of improved river flows.

We will use channel ‘zones’ (Christie and Clark 2002) to frame our hypotheses. Zone A is the portion of the channel that is permanently wetted under baseflow conditions; zone B runs from the edge of zone A to mid-way up the bank; zone C runs from the edge of zone B to the top of the bank (Figure 5.1). Specifically speaking, we expect to see stronger zonation in vegetation types under invariant flow regimes, with terrestrial vegetation encroaching down the river banks, and a very narrow zone of ‘fringing’ vegetation (fluctuation tolerator / fluctuation responder species) (sensu Casanova and Brock, 2000) within zone B (Figure 5.1). In contrast, under more variable flows, the range of fringing species is expected to be much wider within zone B, with the range of obligate aquatic species and terrestrial species accordingly narrower. Terrestrial species would be largely restricted to zone C.

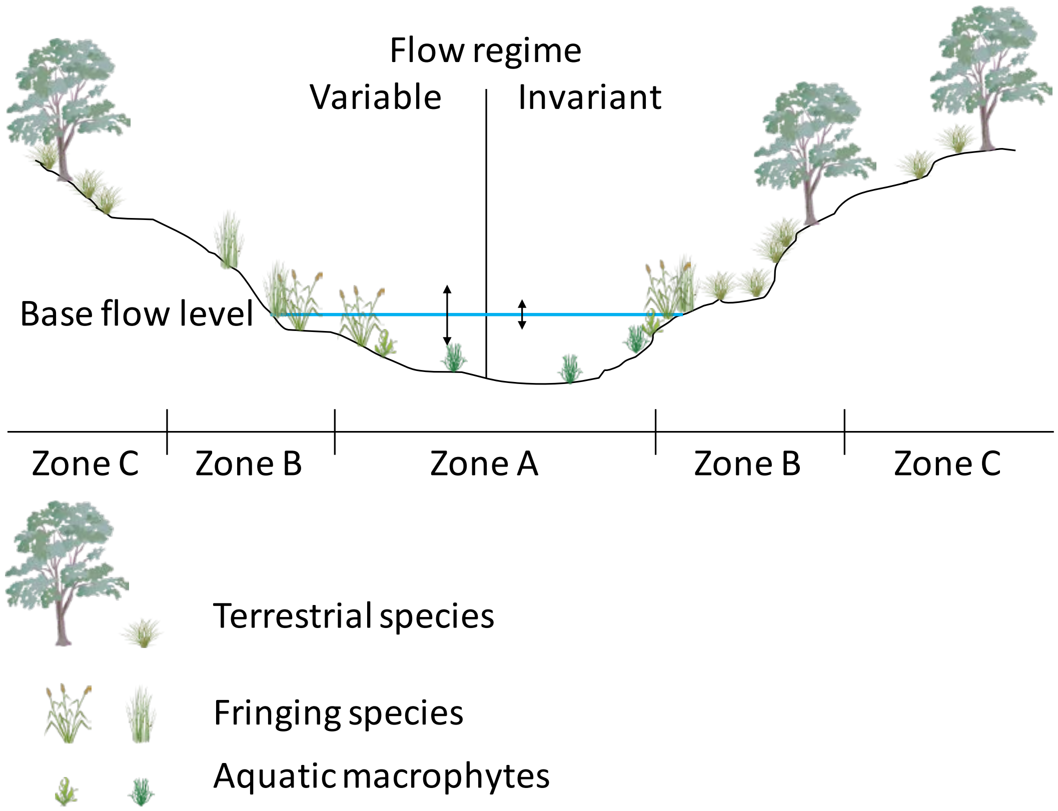


Figure 5.1: Conceptual model of stream vegetation assemblages under varying and invariant flow regimes.

5.4 Methods

Monitoring design

Vegetation monitoring sites on the lower Goulburn River are located at existing VEFMAP (Victorian Environmental Flows Monitoring and Assessment Program) sites (SKM, 2007). We are leveraging the monitoring effort in VEFMAP to provide greater efficiency from CEWH monitoring. Under VEFMAP, vegetation data have been collected at 4 sites: Moss Road, Darcy Track, Loch Garry, McCoys Bridge (Sites 26, 32, 34, 36; Figure 2.1). Those data were collected between 9 and 20 October 2008 (hereafter 2008), 27 January and 3 February 2011 (hereafter 2010), and 9 October and 5 November 2012 (hereafter 2012).

In consultation with the Goulburn-Broken Catchment Management Authority, four new monitoring sites have been established on the lower Broken Creek. Sites are located at Rices Weir, Kennedys Weir, Schiers Weir, and Balls Weir. These four sites were chosen randomly from six potential weir sites; Luckes Weir and Hardings Weir were not chosen. The other two weirs on the lower Broken Creek – Chinamans Weir and Nathalia Town Weir – were excluded from the random selection process because of the close proximity of an artificial levy bank and road, respectively, to the channel. This would have reduced the chances of observing a flow-regulated gradient in vegetation up the river bank.

Field methods

Quadrat monitoring protocols are well established through their use in VEFMAP (e.g. SKM, 2007). In brief, channel cross sections are used to establish the height of permanently-locatable quadrats (so that vegetation can be related to inundation regime), and the vegetation within the quadrat is visually sampled. Cover is assessed using the modified Braun-Blanquet score (Braun-Blanquet, 1964, Cheal and Parkes, 1989) for assessing cover (Table 5.1) VEFMAP data have been collected within the river channel (Zone A), the lower bank (Zone B), or upper bank (Zone C), a delineation that can be used to stratify data for statistical analyses. However, data from Zone B are most important for testing the hypothesis laid out above, and we only collected data from this zone at the new sites in Broken Creek.

Each VEFMAP site has 15 permanently-marked cross-sections, with approximately 10 of these sampled for vegetation (the instruction to contractors is to spend 1 day sampling a site; SKM, 2007). Therefore, at each of the new sites on Broken Creek, we established 10 cross-sections for vegetation sampling. With the focus of the vegetation hypotheses being Zone B, each end of the cross section was located beyond the mid-point of the bank (usually close to or beyond the top of the bank). The cross-sections were located haphazardly moving away from the weir wall, with the 10 cross-sections occupying several hundred metres of the weir pool (Appendix B).

Table 5.1: Modified Braun-Blanquet cover classes. The specified mid-points of the cover classes were used as cover estimates in the statistical analysis.

Category	*Definition*	*Mid-point (%)*
+	Cover less than 1%	0.5
1	Cover 1-5%	3
2	Cover 5-25%	15
3	Cover 25-50%	37.5
4	Cover 50-75%	62.5
5	Cover greater than 75%	87.5

In light of issues with re-locating some VEFMAP vegetation cross-sections for repeat samplings (these issues impinge upon the data analysis in this report – see below), we took the following precautions to ensure that these cross-sections can be located in future years.

i) Each end of each cross-section was marked by a star-picket driven into the ground so that only approximately 5-10 cm was exposed. These stakes will be difficult to remove.

ii) The star-pickets were marked with fluorescent road-marking paint and flagging tape to facilitate re-location (Figure 5.2).

Figure 5.2: Star-picket marked with paint and flagging tape and with cross-section number marked on adjacent tree.

iii) Each star-picket had a key-ring tag attached with the cross-section ID recorded on it (both pickets for each cross-section).

iv) The cross-section number was recorded in road-marking paint on a nearby tree (or large woody debris) to facilitate re-location of the star picket (on both banks) (Figure 5.2)

v) For at least one bank (i.e. it was done for two banks at some sites), we used a GPS-enabled handheld camera to record photo-points from each picket looking a) immediately downwards to record the cross-section number on the key-ring tag, b) downstream and c) across stream (to the other stake) (Figure 5.3).

Figure 5.3: Photopoints recorded for a single cross-section at Balls Weir.

vi) The picket locations were surveyed using a Total Station ® (Figure 5.4), and their locations were mapped to a resolution of a few cm (Appendix B). This will allow re-location of the picket points even if the stakes are removed.

Figure 5.4: Surveying in cross-sections using a Total Station.

As with the Goulburn River sites, vegetation surveys were performed in late spring. At this time of year, plants are easiest to identify in the field because they are flowering. Surveys of the new Broken Creek sites were carried out from 19 -29 November, 2012.

We stretched a 50 m tape measure between the star picket that marked the end of the cross-section and the water's edge. In the highly-regulated environment of Broken Creek, the water surface elevation is virtually constant over time, and was used as a reference point for plotting and analysing data. We surveyed vegetation contained within 1 x 1 m quadrats beginning at the water's edge, and spaced so as to capture major visible changes in vegetation, and also to record at least 4-5 quadrats on each bank of the cross-section (e.g. every 2nd or 3rd metre). The distance along the cross-section and elevation of the upper edge (that furthest from the water) of the quadrat was recorded relative to water level using the tape measure and a dumpy level. We identified all species (to the extent possible in the field) within each quadrat, recording their covers as modified Braun-Blanquet cover classes as described above.

Figure 5.5: Vegetation survey method

Vegetation on the lower Goulburn River and lower Broken Creek were assessed using the same methods, accepting that the use of VEFMAP data from Goulburn River for this study means that data from the two rivers were collected by different personnel, and there would have been some divergence between them in terms of thoroughness and botanical knowledge. We account for these effects as best as possible within the statistical analysis.

All species identified in the field were classified into 'functional groups' (sensu; Casanova and Brock, 2000). Groups within the same functional group are expected to show broadly similar responses to hydrologic regimes. We concentrated on four functional groups, for which species were most abundant (Table 5.2).

Table 5.2: Functional groups included in statistical analysis. Modified from Casanova and Brock (2000)

Functional Group	*Abbreviation*	*Description*
Terrestrial dry	Tdr	Species that germinate, grow and reproduce where there is no surface water and the water table is below the soil surface
Terrestrial damp	Tda	Species that germinate, grow and reproduce on saturated soil
Amphibious fluctuation tolerators, emergent	ATe	Species that germinate in damp or flooded conditions, which tolerate variation in water-level, which grow with their basal portiona under water and reproduce out of the water
Amphibious fluctuation tolerators, low growing	ATl	Species that germinate in damp or flooded conditions, which tolerate variation in water-level, which are low-growing and tolerate complete submersion when water-levels rise

Statistical analysis

The data were analysed using Bayesian models. Bayesian models are inherently flexible, allowing us to fit appropriate model structures to the complex data that often characterise ecological research (Clark, 2005). Bayesian models can also be used predictively – in this study we use the parameterised relationships to predict the likely cover of vegetation on the Goulburn River under different flow regimes. We used hierarchical models for all analyses. Bayesian hierarchical models assume partial dependency of parameter values among sampling units (e.g. sites). One assumes that the model parameters from each sampling unit are drawn from a distribution of plausible parameter values across all sites. This takes advantage of the fact that we can reasonably assume the sites to be similar, and can use this information to improve the power of analyses, but we do not have to assume that the sites are the same, which is clearly unreasonable. The practical effect of the assumption, and of modelling site-level parameters as being drawn from a larger distribution of values, is to reduce the amount of unexplained variation both within and among sampling units (Webb et al., 2010).

We used two different models to compare the bankside vegetation of Broken Creek to the 2012 survey data from the Goulburn River. For these analyses we used elevation of the quadrat above water level as the principal independent variable. The two rivers have very different channel sizes, and so elevations were standardised to the height of the break-of-slope of the bank prior to analysis.

First, we used logistic regression to assess whether the probability of the different functional groups varied with elevation above the invariant water level, and whether this differed between the two rivers. For the hierarchical model, the regression parameters (intercept and slope) at each site were assumed to be drawn from normally-distributed river-level distributions. These analyses were conducted for each of the functional groups. We used the fitted models to calculate the relative elevation (i.e. the proportion of height to break of slope) at which probability of detecting the functional group equaled 0.5.

Second, we conducted analyses of changes in cover with elevation, but did so only on the quadrats that contained higher cover figures at each level of elevation above the water surface. We reasoned that empty (0%) cover of a functional group (of which there were many) may occur for other reasons (e.g. grazing or other unknown drivers) as well as being caused by hydrology. To identify how hydrology limits the distribution of vegetation, one must remove from consideration data that are driven primarily by other mechanisms. To avoid biasing the data set among elevations, we divided the 'sampling effort' into 10 deciles with elevation (i.e. quadrats collected in the lowest 10^{th} of elevations, 10%-20% of elevations, etc.). We then discarded the 50% of quadrats within each decile with the lowest cover scores. Where two or more quadrats had identical cover figures, and one or more was to be deleted, this choice was made randomly. This approach retained low cover quadrats in elevational ranges dominated by such quadrats (and which were therefore likely to be driven by hydrology as well as by other mechanisms), but discarded them in elevational ranges where they were rarer (and therefore likely only to be driven by other mechanisms). This approach is similar in its intent to quantile regression (Koenker 2005), which also seeks to identify the effect of important drivers, given that there are other unknown drivers at work. In particular, both analyses seek to identify the limiting driver (in this case, that which defines the upper limit of vegetation cover), rather than the mean. Following data processing, we plotted the data for each site against elevation, and chose regression functional forms that best characterised changes in abundance with elevation. These were: linear trends for abundance of Terrestrial dry species, linear trends through square-root transformed abundance of terrestrial damp species, exponential trends for abundance of Amphibious fluctuation tolerator emergent species, and no analysis for Amphibious fluctuation tolerator low-growing species, which were virtually absent from the Goulburn River. As with the logistic regressions, regression parameters for each site were assumed to be drawn from river-level normal distributions.

We developed two analyses of Goulburn River vegetation data from 2008, 2010, and 2012, relating vegetation occurrence and cover to the number of days that quadrat had been inundated over the previous 12 months. This choice of inundation period was partly arbitrary, and we did not have the resources to explore other inundation periods within the terms of this project. However, it is worth noting that the 12-month period could be expected to be enough to influence distributions of annual species, of which there were many. The 'hydrological regime' data were provided by ARC project LP100200170 (more specifically, Dr Siobhan de Little, University of Melbourne). The first analysis was analogous to the logistic regressions described above. We related probability of occurrence of the four functional groups to number of days of inundation. The second analysis fitted a non-monotonic function to the data. This was done following preliminary observations that vegetation cover seemed to peak at low (but not zero) number of days of inundation (Appendix C). The function is described by the general form,

$$y = a + b \cdot \ln x + c(\ln x)^2$$

and has the flexibility to show peaks at either extreme or intermediate values, depending on the parameters employed (Mandel, 1981). As with all other models, site level parameters were assumed to be drawn from a river-level distribution.

With this model, we also estimated the marginal effects of environmental flows. We obtained data on environmental flows during the 12 months prior to vegetation sampling in 2012 from the Goulburn-Broken Catchment Management Authority (Geoff Earl, pers. comm.). We used these data, along with the basic flow data at each of the four sites, to compute a hydrograph *in the absence of environmental flows* for each of the four sites. These hydrographs were used to compute new hydrologic regime data (i.e. numbers of days inundated) for the vegetation quadrats at each site. To estimate the marginal effects of environmental flow we compared the modeled cover for each quadrat under actual conditions to the modeled cover in the absence of environmental flows.

5.5 Results

Description of field data

At Broken Creek, we surveyed 373 quadrats, identifying 108 unique species (Table 5.3). A complete species list is provided in Appendix A in Table A.1. We expended a disproportionate amount of effort at the first site sampled (Rices Weir) before reducing survey effort for the other sites to better match the sampling effort at VEFMAP sites.

Anecdotally, the data for the lower Broken Creek appear to support our prediction that vegetation assemblages in this flow-invariant environment are marked by strong zonation of vegetation types. For many cross-sections there was a distinct zone of fringing aquatic vegetation at the water's edge (e.g. Figure 5.7). Indeed for some cross sections, aquatic vegetation at the water's edge had grown to the point where it had compacted and consolidated providing a substrate upon which we were able to walk. Beyond these reasonably narrow zones, vegetation was generally much more sparse, and characterised by grasses and other terrestrial species, with fluctuation tolerator species (e.g. Juncus sp.) also present, but much less common than at the water's edge (Figure 5.8).

Table 5.3: Summary of lower Broken Creek surveys

	Number of quadrats	*Number of species*
Rices Weir	141	69
Kennedys Weir	69	54
Schiers Weir	77	51
Balls Weir	86	48

Figure 5.6: River bank at Schiers Weir showing a very high coverage of pasture grasses near the upper end of the cross-section.

VEFMAP data on the Goulburn River found a total of 263 species across the three years of sampling (Table 5.4). A complete species list is provided in Appendix A in Table A.2.

Table 5.4: Summary of Goulburn River VEFMAP surveys

	Number of quadrats in zone B	*Number of species*
Moss Rd	70 (2008), 52 (2010), 54 (2012)	132
Darcy Track	93 (2008), 52 (2010), 64 (2012)	138
Loch Garry	73 (2008), 51 (2010), 68 (2012)	123
McCoys Bridge	57 (2008), 34 (2010), 44 (2012)	139

Figure 5.7: River bank at Kennedys Weir showing a distinct zone of fringing vegetation near the non-varying water's edge.

Figure 5.8. Vegetation density reduces with distance from the water's edge. Fluctuation-tolerant species such as *Juncus* sp. are found in both areas, but are much less common away from the water's edge.

Comparisons of Broken Creek vegetation data to those of Goulburn River

Exploratory data analysis revealed some pattern in zonation of the different functional groups among the four sites on each river, but no overwhelmingly obvious differences (Appendix C, Figure C.1 to C.4).

However, statistical analyses revealed strong patterns in terms of both occurrence and cover. For the logistic regression, probability of occurrence of Terrestrial dry species increased with increasing elevation above the water surface at all sites (Figure 5.9). However, sites on Broken Creek had very high probabilities of occurrence at very low elevations. We conclude that the invariant water levels on the Broken Creek have increased the habitable range for Terrestrial dry species, in line with predictions of the conceptual model.

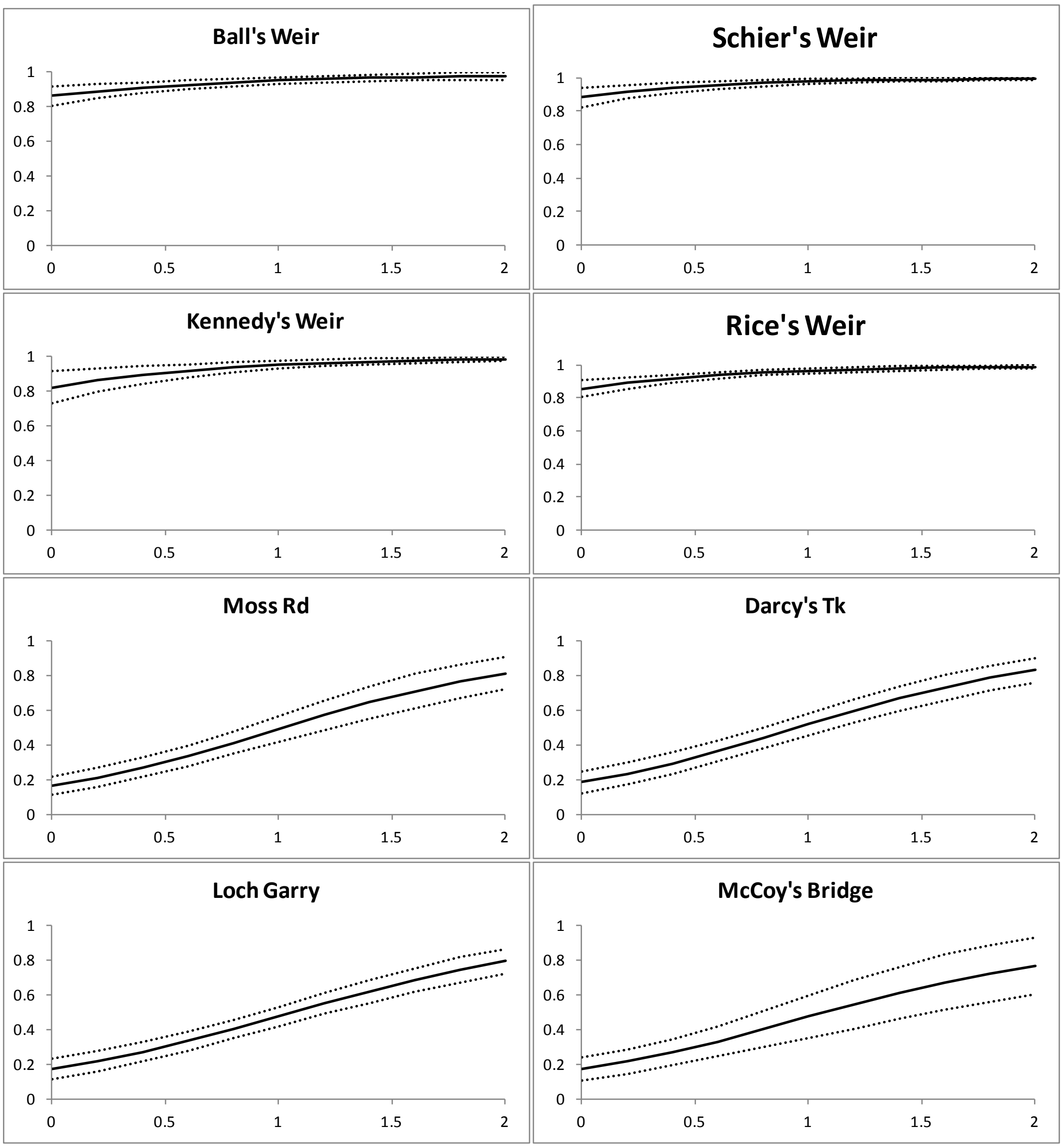


Figure 5.9: Probability of occurrence of Terrestrial dry species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Solid lines are mean estimated probability of occurrence. Dotted lines are ± 1 SD of the estimate.

When examining the ‘top 50%’ of the data (i.e. with the most cover) and analysing abundance, the difference between the two rivers disappears. All sites show increasing abundance with elevation, and there are no patterns to distinguish one river from the other (Figure 5.10).

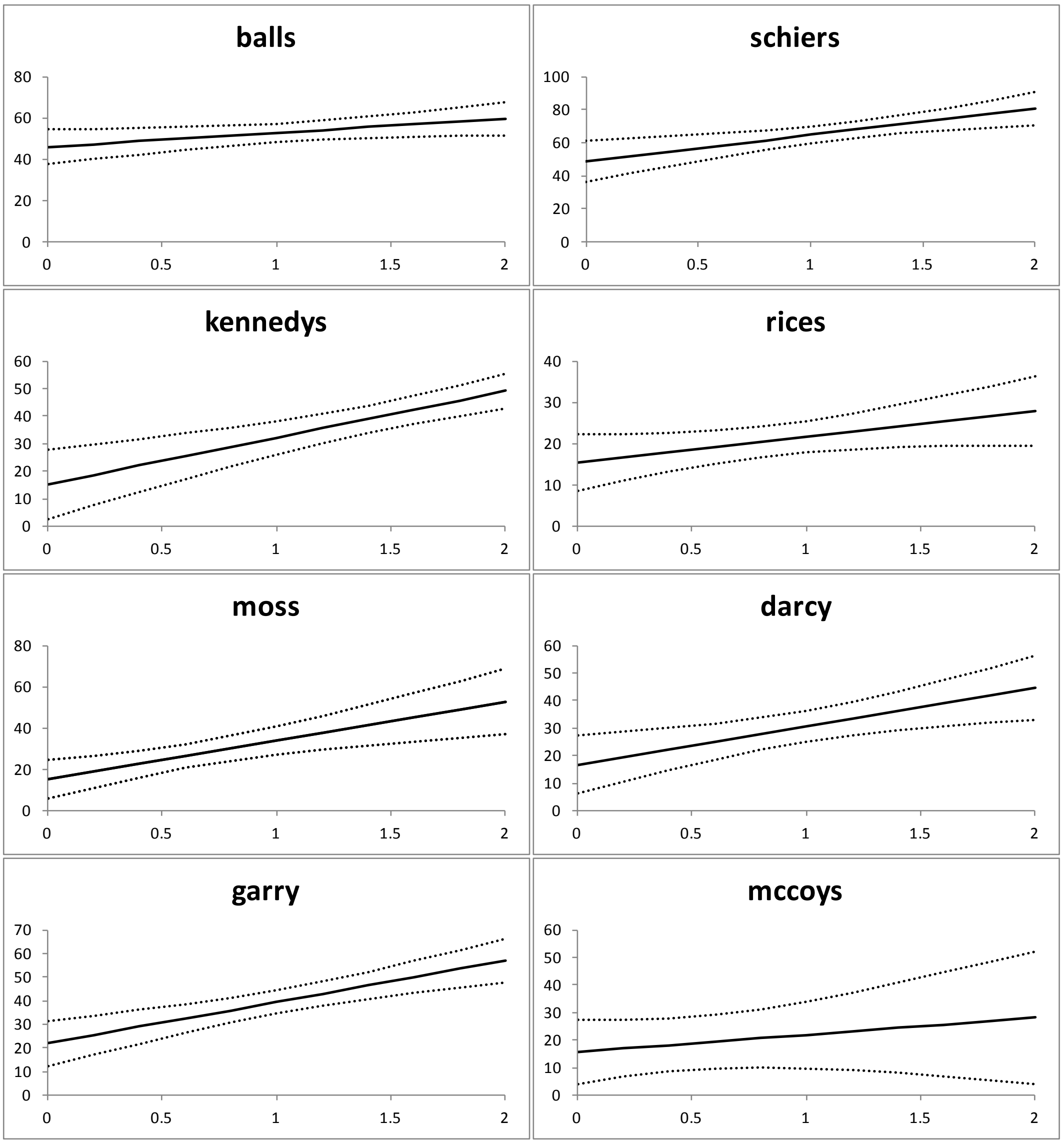


Figure 5.10: Cover of Terrestrial dry species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Solid lines are mean estimated percentage cover. Dotted lines are ± 1 SD of the estimate.

The pattern was different for Terrestrial damp species (Figure 5.11). While sites on Goulburn River exhibited an increasing probability of occurrence with increasing elevation, the opposite pattern was seen at Broken Creek; Terrestrial damp species were more likely to be found closer to the water's edge. We believe this pattern has come about because the highly regulated conditions at Broken Creek create a zone of damp soil very close to the water's edge that is rarely flooded. This provides ideal conditions for Terrestrial damp species. In contrast, the more variable inundation heights on the Goulburn make Terrestrial damp species less likely to occur immediately adjacent to the water's edge.

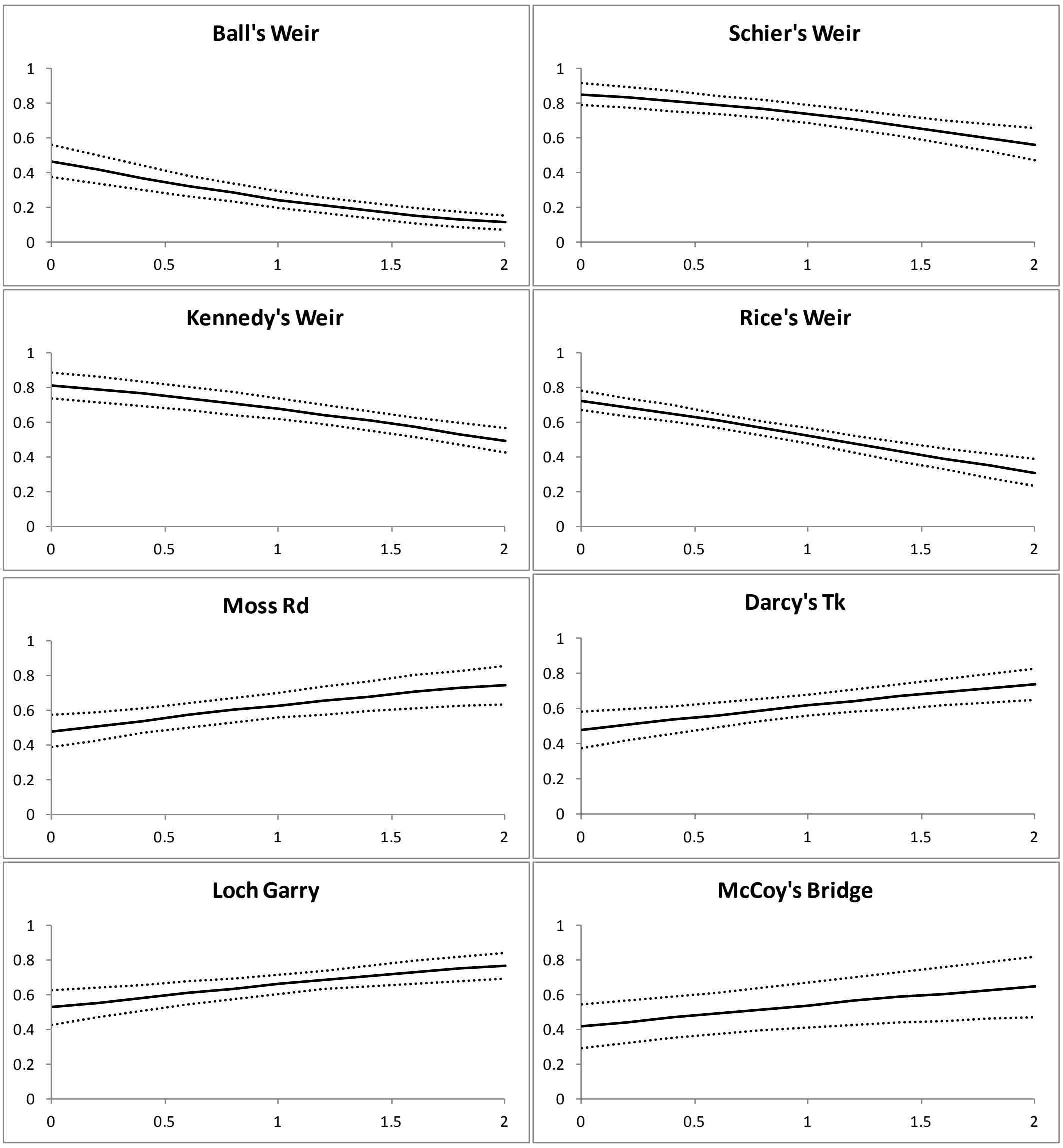


Figure 5.11: Probability of occurrence of Terrestrial damp species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Solid lines are mean estimated probability of occurrence. Dotted lines are ± 1 SD of the estimate.

This pattern is, for the most part, recapitulated by the analysis of the top 50% of cover quadrats (Figure 5.12). For all the Goulburn River sites, cover increases with increasing distance from the water surface. The opposite is seen for all of the Broken Creek sites except for Schiers Weir; cover is highest immediately adjacent to the water surface. This effect is particularly strong at Balls Weir and Rices Weir, with close to zero abundance above 1.5 x the break of slope height. We have no plausible explanation for the weak increase in abundance at Schiers Weir, given that the hydrological regime at that site is almost identical to that at the other Broken Creek sites. However, the effect is not significant, and we have not attempted to interpret it further.

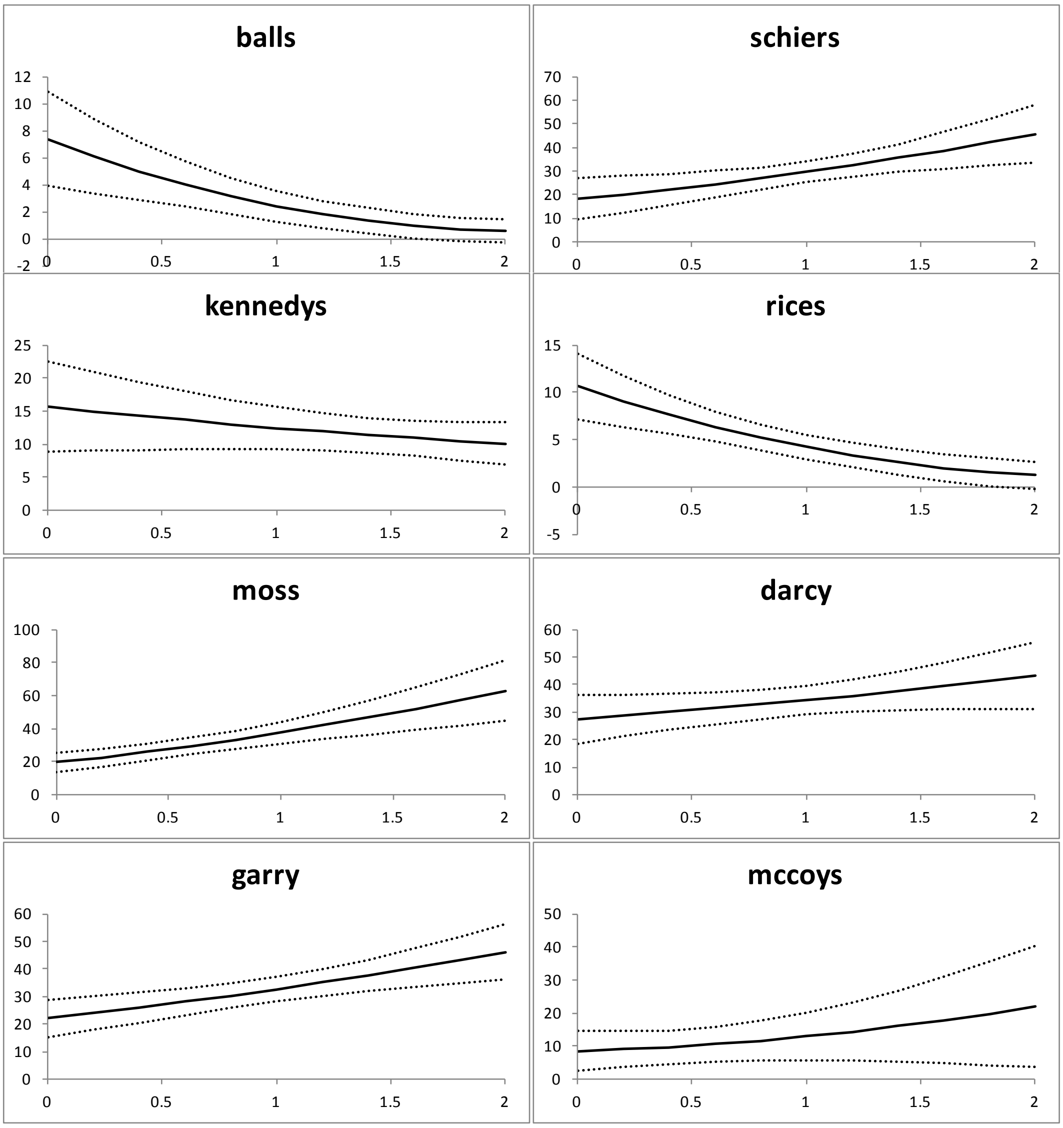


Figure 5.12: Cover of Terrestrial damp species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Solid lines are mean estimated percentage cover. Dotted lines are ± 1 SD of the estimate.

Occurrence of Amphibious fluctuation tolerator, emergent species shows functionally similar patterns to Terrestrial damp species, but the patterns were much stronger in Broken Creek in particular. At Kennedys Weir, Rices Weir and Schiers weir probability of occurrence of these species is almost 1 at the water's edge and drops to zero by approximately 1.5 x the break of slope height (Figure 5.13). This is the sort of strong zonation expected under flow-invariant conditions by our conceptual model. The pattern is weaker at Balls Weir, but there is still a negative relationship with elevation. In contrast, the four Goulburn River sites show weak positive relationships of occurrence with elevation. This pattern runs contrary to our conceptual model, but can be explained in light of the recent flooding in the Goulburn River.

Fluctuation tolerator species, once established, can withstand considerable emersion. Therefore it is not unreasonable to find these species well up the bank on the Goulburn River. The 2010 and 2012 floods scoured the river banks (G. Earl, GBCMA pers. comm.), stripping them of much of their vegetation. Such scouring would have been more severe lower down the bank, where inundation periods would have been longer and shear stresses higher. Thus, the pattern observed for fluctuation tolerator emergent species (and indeed the other functional groups as well) could well be caused by the removal of vegetation during flooding, rather than by preferential growth at different levels of inundation.

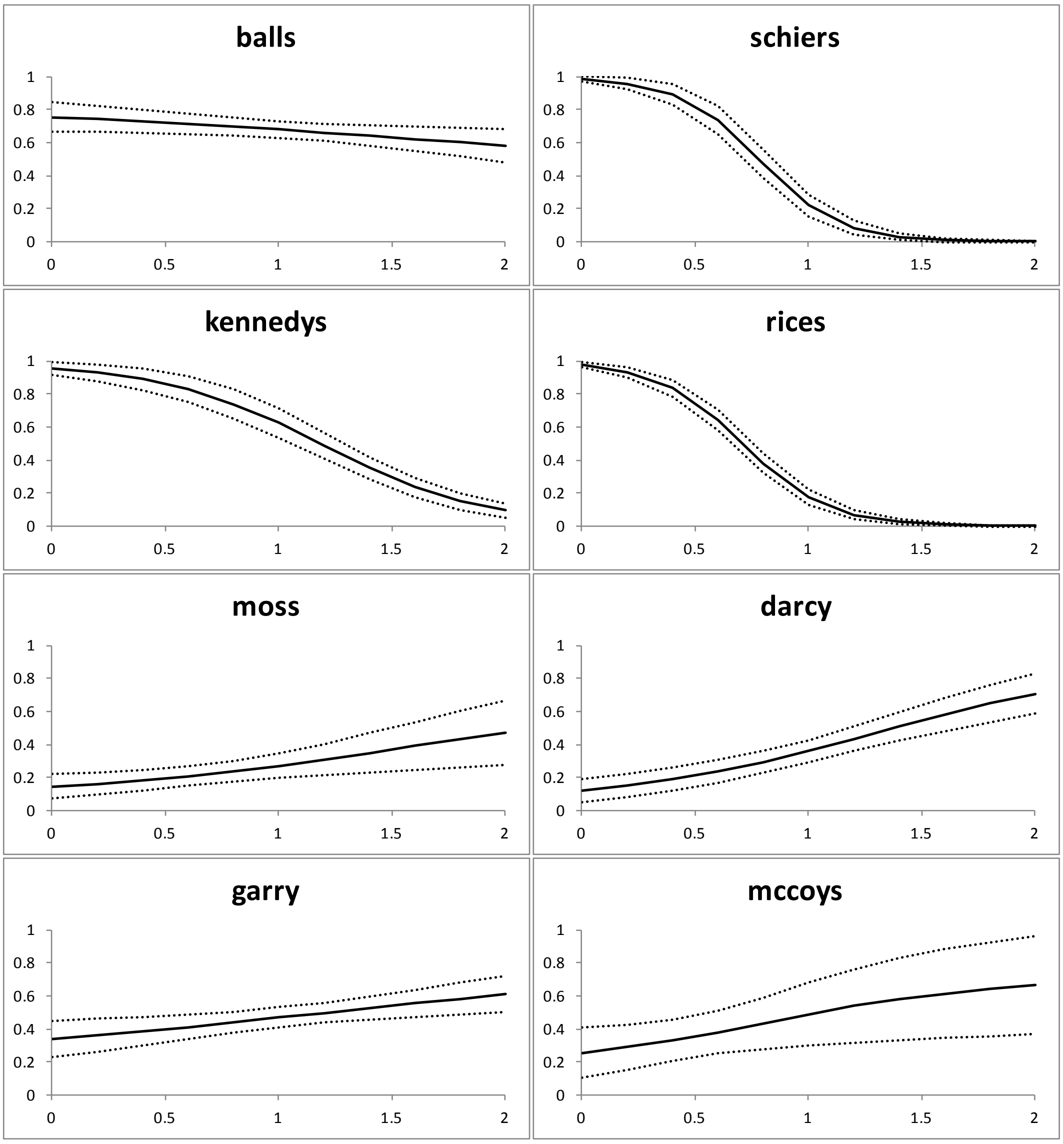


Figure 5.13: Probability of occurrence of Amphibious fluctuation tolerator, emergent species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Solid lines are mean estimated probability of occurrence. Dotted lines are ± 1 SD of the estimate.

The exponential regression of cover of the top 50% of cover quadrats reconfirm these findings (Figure 5.14). Covers are high (but variable) close to the water surface, but drop away to near zero at approximately 1.5 x the break of slope height on the Broken Creek sites (with the exception of Schiers Weir). In contrast, covers increase slightly with elevation on the Goulburn River. However, the pattern of increase is weaker than was the case for probability of occurrence.

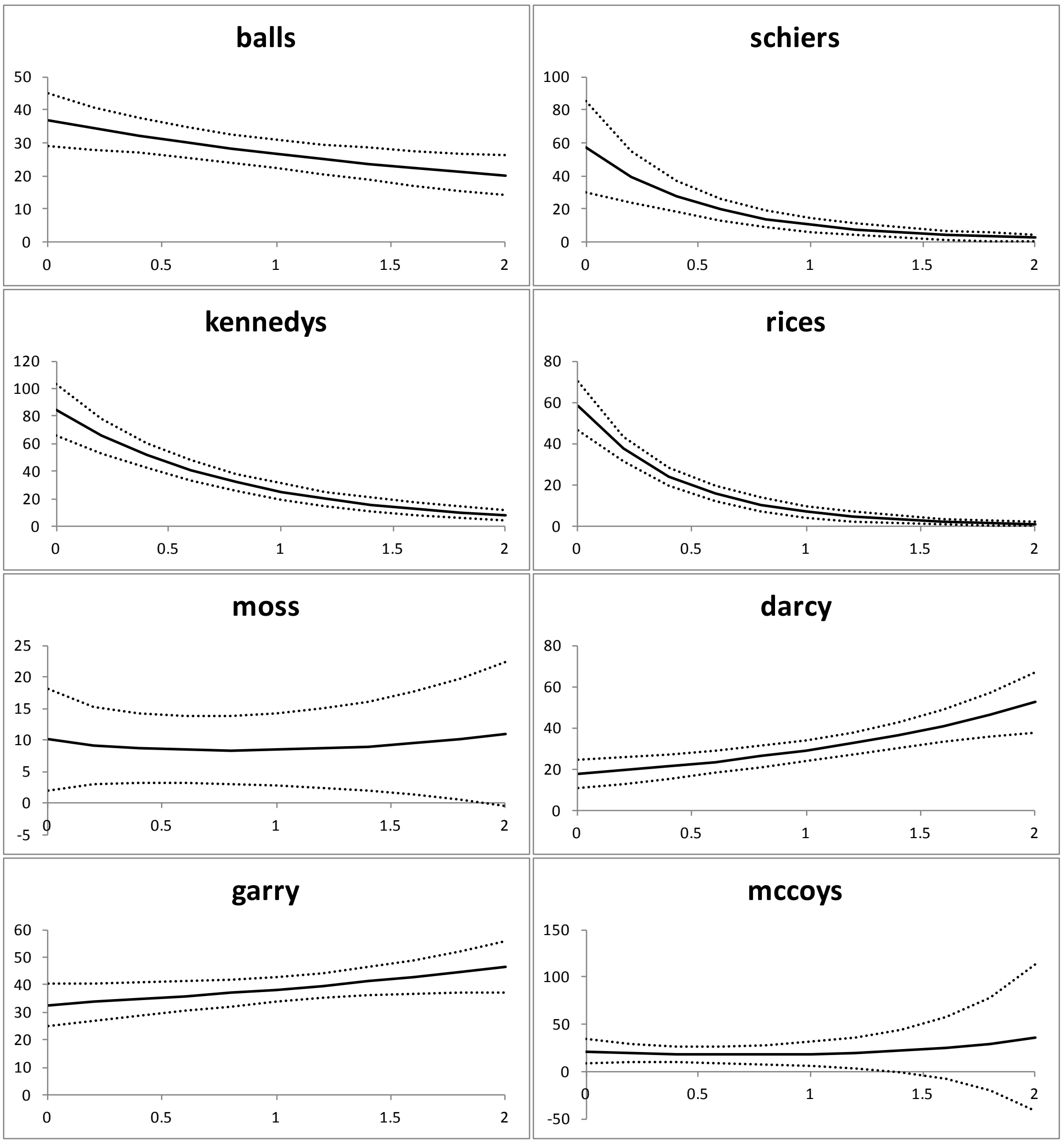


Figure 5.14: Cover of Amphibious fluctuation tolerator, emergent species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Solid lines are mean estimated percentage cover. Dotted lines are ± 1 SD of the estimate.

Finally, Amphibious fluctuation tolerator, low-growing species were almost absent from the Goulburn 2012 data set. As such, no pattern in their distribution can be discerned, and we do not present any analysis of those data. In contrast, this functional group exhibits a similarly strong zonation in occurrence at Broken Creek sites as do emergent Amphibious species (Figure 5.15),

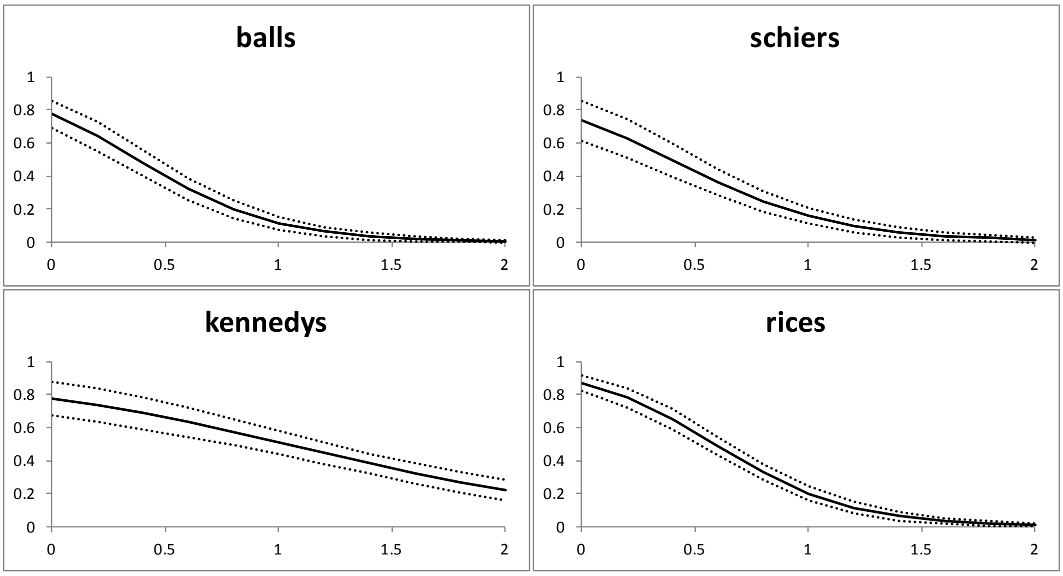


Figure 5.15: Probability of occurrence of Amphibious fluctuation tolerator, low-growing species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek. Solid lines are mean estimated probability of occurrence. Dotted lines are ± 1 SD of the estimate.

Summary of results for comparison between Broken Creek and Goulburn River

In general, the results conform to our prior expectations as to the effects of an invariant versus variable flow regime on vegetation. Strong zonation in the amphibious species demonstrates that Broken Creek has a narrow optimal range for these species. Patterns for Terrestrial damp species were especially interesting, with the highly regulated conditions of Broken Creek effectively reversing the normal gradient of this species (more common close to the water's edge). Overall, these results give an indication of the effects of strong flow regulation upon bankside vegetation. However, this does not provide a prediction of the effects of environmental flows *per se*.

Relationships of vegetation to hydrological regime in the Goulburn River

Exploratory data analysis found no obvious differences between years of sampling or number of inundation events (separate to effects of inundation). We therefore concentrated on inundation period as the main independent variable. Much of the time, it appeared that maximal abundances occurred with short (but not zero) inundation periods (Appendix C: Figure C.5 to C.8).

All vegetation functional groups showed reduced probability of abundance with increasing duration of inundation. This is not surprising, as none of the four are aquatic specialists. The pattern was consistent among sites, although some sites supported greater abundances (and therefore had a higher probability of occurrence) than others. Probability of occurrence of terrestrial species was high under low rates of inundation (Figure 5.16, Figure 5.17), but fell away to zero under constant inundation. Indeed, it ought to be impossible to have any terrestrial vegetation cover under near constant inundation. We believe the non-zero effects shown in these plots reflect errors in the matching of vegetation survey data to separately-conducted cross-sectional transect surveys that supply the elevation (and hence inundation) data used in these analyses (S. de Little, University of Melbourne, pers. comm.).

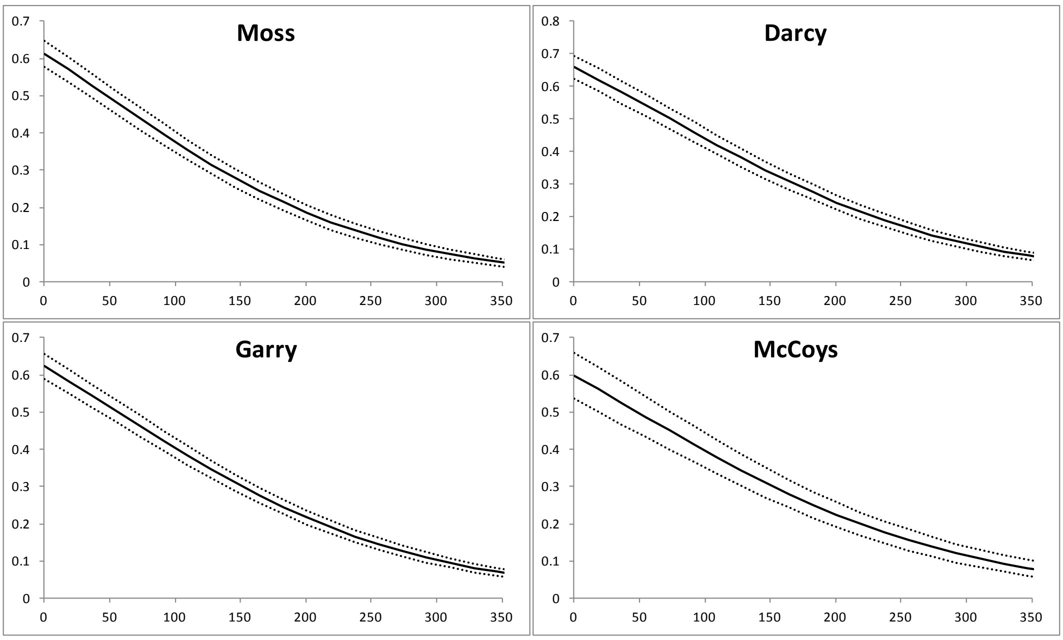


Figure 5.16: Probability of occurrence of Terrestrial dry species (y-axis) with number of days inundated per year (x-axis) for sites on the Goulburn River. Solid lines are mean estimated probability of occurrence. Dotted lines are ± 1 SD of the estimate.

Probabilities of occurrence of the two amphibious functional groups are much lower, reflecting lower abundance in the survey data of these taxa. Nevertheless, they too show further reduced probability of occurrence under increasing inundation (Figure 5.18, Figure 5.19).

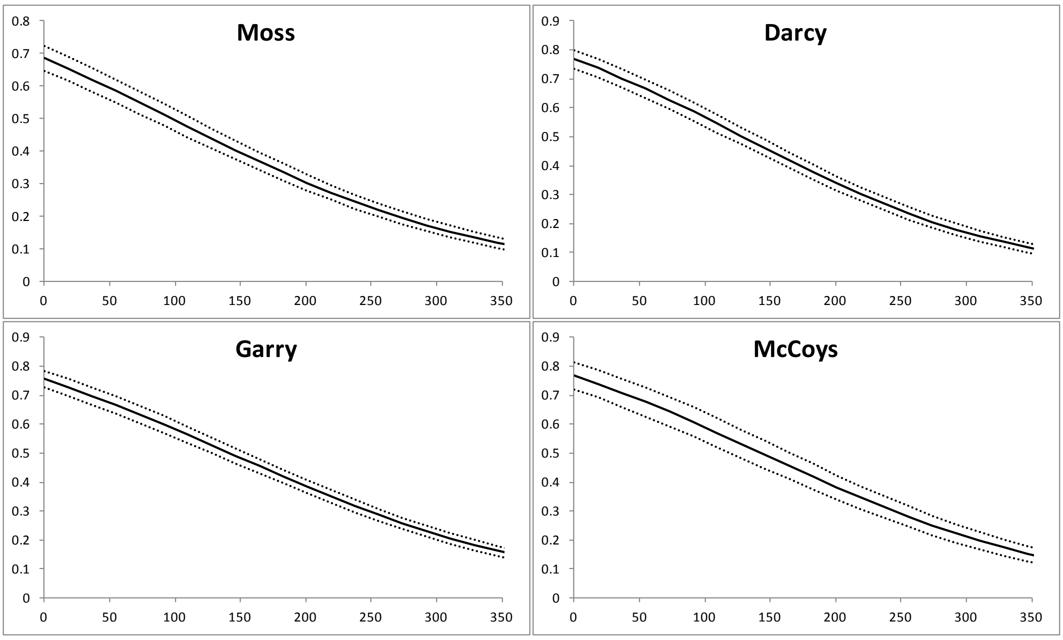


Figure 5.17: Probability of occurrence of Terrestrial damp species (y-axis) with number of days inundated per year (x-axis) for sites on the Goulburn River. Solid lines are mean estimated probability of occurrence. Dotted lines are ± 1 SD of the estimate.

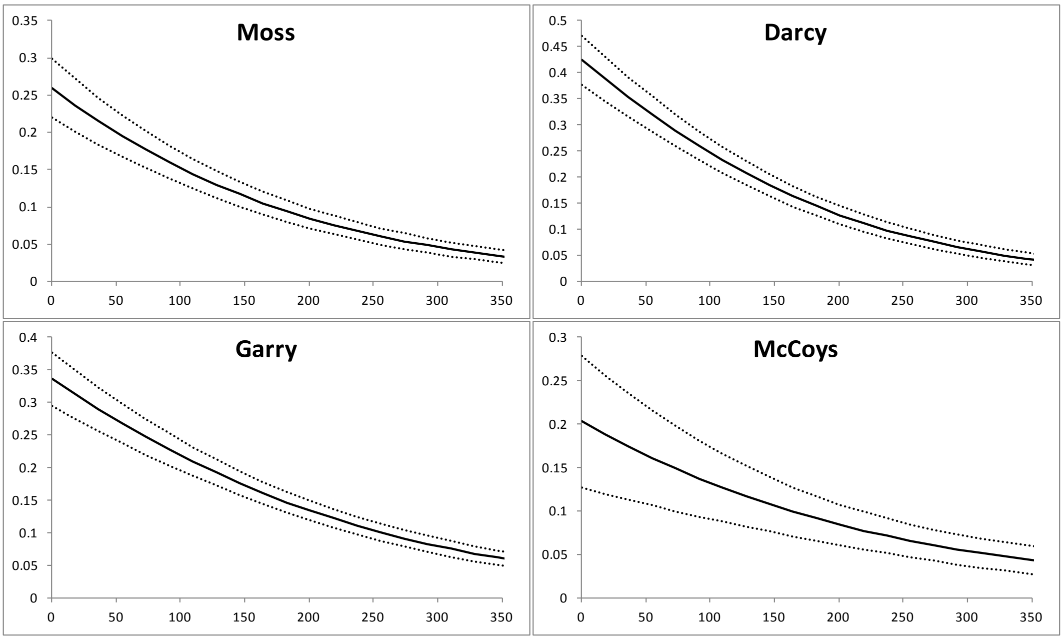


Figure 5.18: Probability of occurrence of Amphibious fluctuation tolerator, emergent species (y-axis) with number of days inundated per year (x-axis) for sites on the Goulburn River. Solid lines are mean estimated probability of occurrence. Dotted lines are ± 1 SD of the estimate.

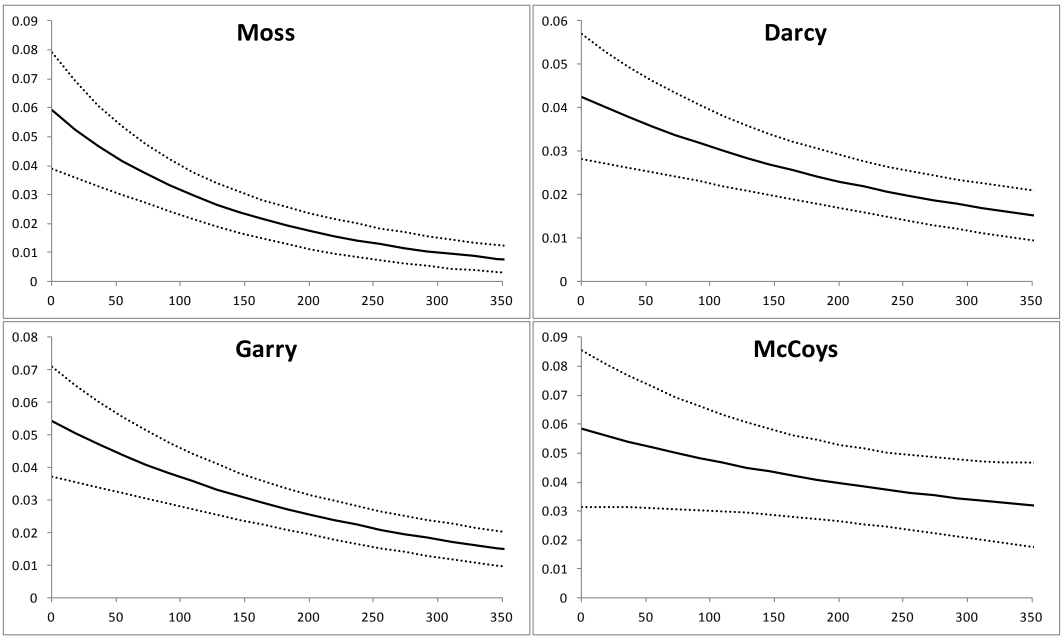


Figure 5.19: Probability of occurrence of Amphibious fluctuation tolerator, low-growing species (y-axis) with number of days inundated per year (x-axis) for sites on the Goulburn River. Solid lines are mean estimated probability of occurrence. Dotted lines are ± 1 SD of the estimate.

The abundance of Terrestrial dry species follows a virtually monotonic relationship with increasing duration of inundation (Figure 5.20). This is consistent with the conceptual model, and with the observations from Broken Creek. Simply, Terrestrial dry species are disadvantaged by more than a small amount of inundation.

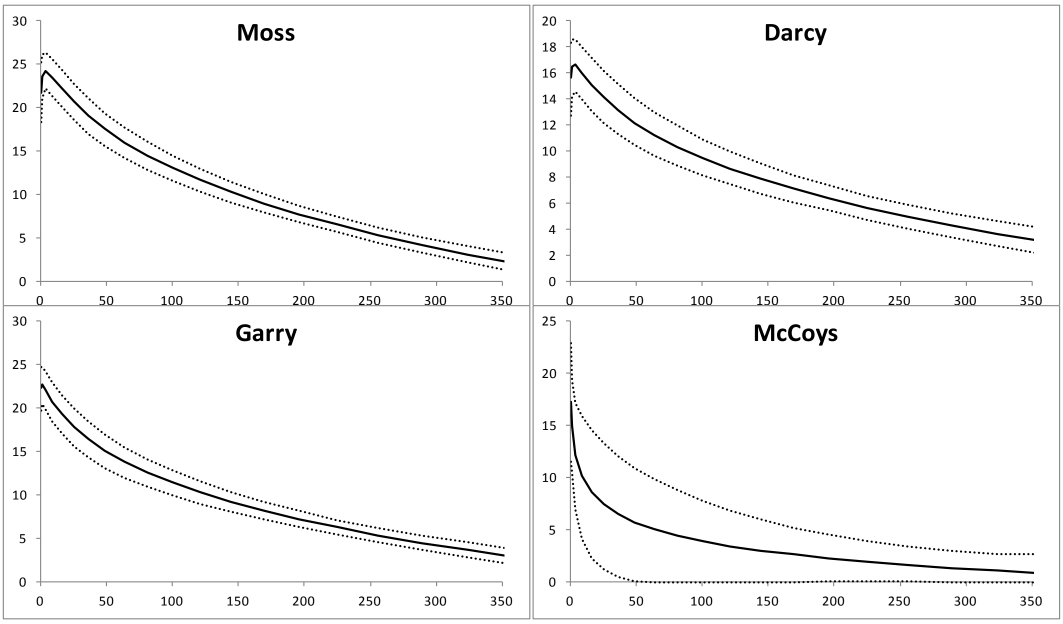


Figure 5.20: Cover of Terrestrial dry species (y-axis) with number of days inundated per year (x-axis) for sites on the Goulburn River. Solid lines are median estimated abundance. Dotted lines enclose the 95% credible interval of the estimate.

The pattern for Terrestrial damp (Figure 5.21) and Amphibious fluctuation tolerator, emergent (Figure 5.22) species is slightly different, with peak abundances occurring at low number of days of inundation, rather than at zero. This reflects the original observations during exploratory data analysis, and is consistent with the conceptual model to the extent that both functional groups should be advantaged by some inundation. It is also consistent with observations from Broken Creek, where abundance of both these groups was highest near the water level. It is slightly surprising that the peak abundance for Amphibious fluctuation tolerator, emergent species does not occur at a higher number of days compared to Terrestrial damp species. Our expectation of life history adaptation would suggest they could cope with longer inundations than Terrestrial damp species.

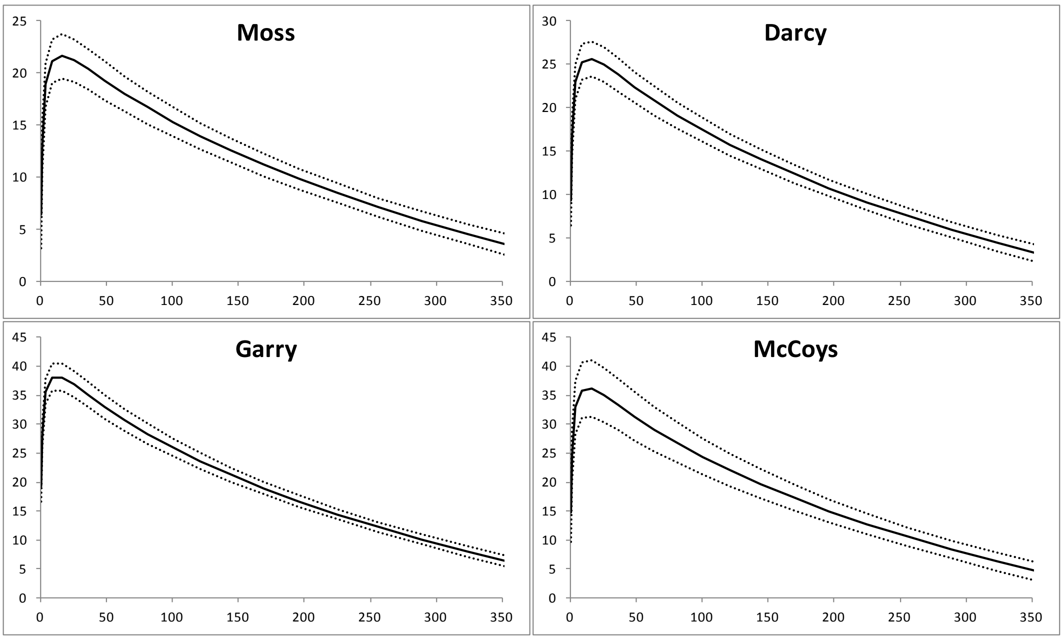


Figure 5.21: Cover of Terrestrial damp species (y-axis) with number of days inundated per year (x-axis) for sites on the Goulburn River. Solid lines are median estimated abundance. Dotted lines enclose the 95% credible interval of the estimate.

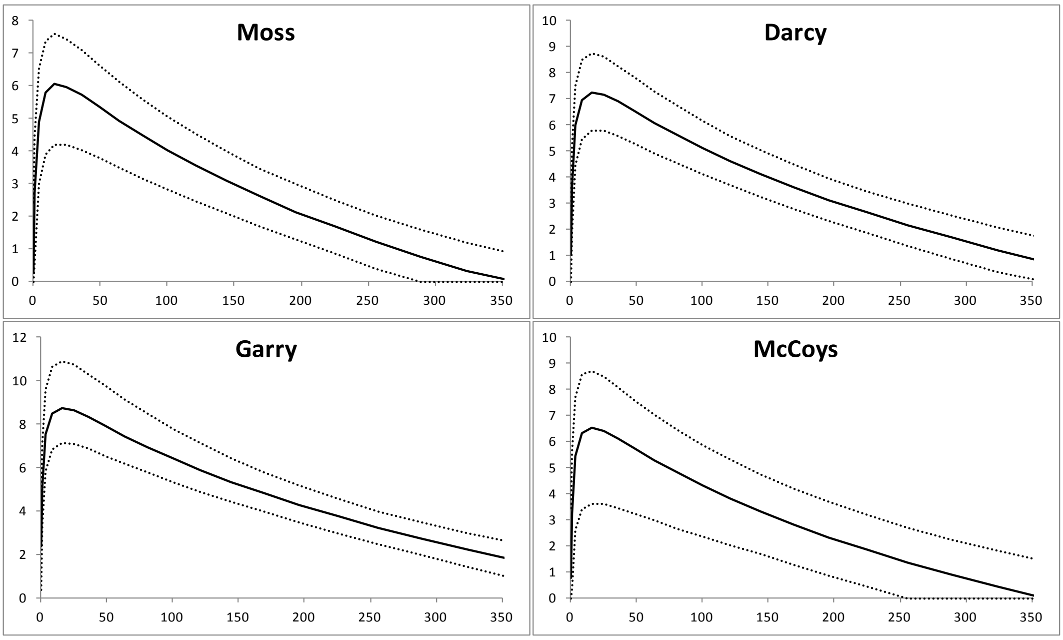


Figure 5.22: Cover of Amphibious fluctuation tolerator, emergent species (y-axis) with number of days inundated per year (x-axis) for sites on the Goulburn River. Solid lines are median estimated abundance. Dotted lines enclose the 95% credible interval of the estimate.

The very low abundance of Amphibious fluctuation tolerator, low-growing species precluded the opportunity for any analysis.

Benefits of environmental flows

The plots presented above give predictions of expected cover under different inundation durations. However, the width of the x-axes (a full year) is far beyond the range of conditions likely to be achieved with environmental flows. To provide a better indication of what can be achieved through environmental flow delivery, we modeled expected abundance of the three functional groups above at the quadrat level. Overall effects are very small (Table 5.5).

Table 5.5: Change in average predicted percentage cover of three functional groups of plants as a result of removing environmental flows from the Goulburn River. The differences are extremely small because results are averaged across all quadrats.

Functional group	*Average cover (%)*	*Average cover with no environmental flows (%)*	*Proportional increase*
Tdr	2.19	2.49	0.14
Tda	23.47	24.29	0.03
ATe	2.55	2.86	0.12

This is to be expected as the removal of environmental flows from the flow regime has no effect on the inundation histories of 15% of the quadrats (92 out of 633 modelled across the four sites), and only minor effects on the inundation regimes of many other quadrats. A more ecologically meaningful comparison is for quadrats for which the removal of environmental flows would lead to a substantial change in inundation regime.

Such quadrats are those found at the margins of the river, Approximately 20 quadrats suffered either or both of a > 20 % reduction in the number of days inundated per year, or an absolute reduction of 70 or more days when environmental flows were removed from the flow regime. Such differences can be expected to have effects for those quadrats.

Table 5.6 shows quadrats for which substantial changes were seen. Although the absolute percentage changes of the vegetation functional groups are still small, these changes can represent up to a several hundred percent increase in the cover of the quadrat compared to what is expected under the environmental flow regime. These differences are also much larger than the prediction uncertainty of the model (not shown). Thus we can be confident that the flows are having an effect.

5.6 Discussion

Have environmental flows promoted the development of flood-tolerant vegetation on the banks of the Goulburn River?

The monitoring and analysis presented here shows universally negative relationships of vegetation cover with inundation. It is only at the very lowest levels of inundation that increased inundation is expected to lead to increased abundances of Terrestrial damp and Amphibious fluctuation tolerator, emergent species.

The establishment of new sites on the relatively flow-invariant Broken Creek provides a point of comparison against which we can assess vegetation data from the lower Goulburn River, which receives environmental flows, including Commonwealth environmental water. We expended considerable effort ensuring that the cross sections at these sites will be re-locatable in future; this will allow us to assess change at these sites over time in relation to that experienced at the more flow-variable sites on the Goulburn River.

These sites on Broken Creek give us the most extreme example of what could happen under a highly regulated flow regime. There, terrestrial vegetation covers the entire bank down to the water surface. In contrast, the Goulburn River, with its more variable flows, supports little vegetation. However, it is arguable that this represents a more natural state, and this is backed up by the modeling.

There remains some uncertainty as to whether the lower Goulburn River is currently recovering from the 2010 and 2012 floods. Anecdotal information suggests that the floods removed much of the aquatic vegetation from the lower Goulburn, and that it has yet to re-establish (T. Barlow, GBCMA, pers. comm.). The same is also possible for the amphibious taxa analyzed in this project. However, the vegetation patterns in 2010 and 2012 (both surveys done post-flood) are not markedly different to those taken during 2008, an historically dry period. It could be said that none of the data has been collected during ‘average’ conditions for the Goulburn, which suggests that repeating this monitoring in 2013/14 will be valuable for further improving our understanding.

Table 5.6: Twenty of the quadrats most affected by the modeled removal of environmental flows. Columns show original modeled cover under the existing environmental flows regime and the increase in cover that would have occurred if the flows were not delivered. All figures are % of quadrat area.

Tdr		*Tda*		*ATl*	
Original	*Change*	*Original*	*Change*	*Original*	*Change*
1.94	9.60	3.16	10.66	0	3.51
10.76	4.42	13.02	4.3	3.24	1.46
0	0	32.99	3.78	0.09	1.71
6.46	1.50	29.43	2.69	2.26	1.04
2.30	2.29	3.59	2.67	0	0.91
2.30	2.29	3.59	2.67	0	0.91
2.30	2.29	3.59	2.67	0	0.91
2.33	2.26	3.63	2.63	0.01	0.91
2.33	2.26	3.63	2.63	0.01	0.91
2.27	2.22	3.56	2.58	0	0.87
2.27	2.12	3.56	2.47	0	0.83
2.27	2.12	3.56	2.47	0	0.83
2.27	2.12	3.56	2.47	0	0.83
0	0	13.58	2.42	1.43	0.95
0	0	13.55	2.38	1.41	0.94
0	0	34.29	2.3	0.67	1.04
7.61	1.36	31.51	2.27	3.06	0.89
0	0	13.55	2.24	1.41	0.88
2.27	1.89	3.56	2.20	0	0.74
2.27	1.86	3.56	2.16	0	0.73

The analyses do not permit us to make direct predictions of how long vegetation assemblages would take to respond to the instigation of an environmental flow regime, but some inference on this topic is possible. The three seasons of sampling data on the Goulburn River are repeated measures of the same quadrats. Preliminary examination of the data suggested that there was little temporal autocorrelation among repeated measures of the same quadrats (results not shown), suggesting that vegetation cover changed considerably among the three sampling seasons. Such dynamic behavior of vegetation cover is most likely due to changes in the cover of annual species; long-lived perennial species can be expected to change much more slowly. Therefore, it is reasonable to expect reasonably rapid changes in streamside vegetation (~ 1-2 years) following the instigation of environmental flows, but only for those portions of the stream bank whose inundation regime is substantially affected by the changed flow regime.

In summary therefore, we have described major differences between the flora of the Goulburn River and Broken Creek, with at least some of this difference being caused by the different levels of flow regulation among the two systems. The predictions of our conceptual model were borne out, with one or two exceptions, and the monitoring shows the magnitude of change in vegetation assemblage that is possible. On the Goulburn River, we were able to quantify relationships between inundation and both occurrence and cover, and to use these relationships to model the effects of environmental flows in the Goulburn in 2012. While these effects were small, they were larger than the prediction uncertainty of our model, giving us faith in the result. More importantly, they were substantial for the individual quadrats most affected.

6 FISH AND MACROINVERTEBRATE PHYSICAL HABITATS

6.1 Evaluation question and expected outcomes

This section addresses the evaluation question: Have environmental flows improved physical habitat conditions for fish and macroinvertebrates? The expected outcome was that minimum environmental flows would increase deep water habitat availability but there was a concern it may reduce slackwater habitats. In addition, environmental flows delivered as freshes was expected to flush fine sediments that had been deposited on the streambed during baseflows and improve physical habitat conditions on and in the streambed.

6.2 Key findings

- Slackwater habitat (slow and shallow water) is increased in abundance by baseflows and reduced by flow freshes
- Deepwater habitat (pools > 1 m) is increased in abundance by environmental flows
- Sediment smothering is highly variable spatially within the channel, and the response to flow is dependent on available sediment concentrations. Environmental flows can both exacerbate sediment smothering (following large watershed-driven bankfull events) but also reduce the potential for sediment smothering in the latter parts of the irrigation season

6.3 Background

In addressing the question *"Have environmental flows improved physical habitat conditions for fish and macroinvertebrates?"* we focus on the following:

1. Slackwater habitat
2. Deepwater habitat
3. Substrate sediment condition

Physical habitat

These physical habitats have been chosen based on empirical associations to fish and macroinvertebrate richness and diversity, and their sensitivity to changes in the hydrologic regime. Physical habitats are the result of interactions between the morphology of the channel and the flow through them, resulting in variations in velocity and depth that favour particular taxa. These conditions can be critical during many stages of life for fish and macroinvertebrates.

Slackwaters - slow flowing and shallow habitat - are important refuges and productive zones, particularly during early life stages for larval and juvenile fish (Schiemer et al. 2001, Humphries et al. 2006). They increase flow residence time increasing opportunities for retention of solutes through biological uptake, suspended particulate matter and juvenile and larval fish (Price 2007, Vietz et al. 2013).

Flow regulation alters the number, abundance and distribution of slackwaters as they are sensitive to changes in discharge. Vietz et al. (2013) found that for increases in discharge above baseflows slackwaters generally contracted to the channel margins as smaller, dispersed habitats, and in a heavily regulated system slackwaters were reduced by more than 50%, and patch sizes decreased 2 to 5 times. Baseflows tend to provide greater slackwater abundance than very low flows, however, as demonstrated by Vietz et al. (2013), higher discharges (e.g. prolonged high flows, as well as freshes) can reduce slackwater abundance.

Deepwater habitat, or pools, are known to provide important physical habitat as refuges for fish (King et al. 2009). Baseflows tend to improve the availability of deepwater habitat during otherwise low flow periods.

Quantifying and understanding the relationship between environmental flows and these habitats can assist in reducing the impacts of regulation and informing flow management, particularly with regard to the importance of timing and seasonality of physical habitat.

Substrate sediment condition

The surficial and pore spaces (interstices) of bedload substrates such as sands and gravels provide an important physical habitat, particularly for fish and macroinvertebrates (Koehn et al., 1994, Price 2007). These substrates, however, are often ‘smothered’ by fine-grained sediments. These fine-grained sediments - composed of silt and clay-sized particles in the Goulburn River (usually no greater than 100 μm in diameter) - are transported as suspended sediment.

During low flow conditions these sediments settle out of suspension forming ephemeral deposits within the mobile substrate, and within the interstitial pore spaces, smothering the riverbed and reducing the suitability of the habitat. Sediment smothering can increase drift, reduce respiration and feeding activity, smother eggs and entomb fry (Reiser et al. 1998, Harrison et al. 2007). With sediment smothering we are concerned that fine-grained sediments (silts/clays) infill the interstices, and smother the coarser bed sediments (sands/gravels).

Low water levels, or regulated flows maintained at consistent water levels, when velocities are low, can lead to sediment smothering. Sediment smothering can also be increased for higher concentrations of fine-grained sediment (suspended sediment). Larger flows causing erosion can lead to higher concentrations, as can other activities such as land clearing, boating etc. Environmental flow freshes, or flow variations, are commonly delivered to alleviate this issue (through scouring). The complexities of relationships between flows and sediment smothering may explain why this doesn’t seem to have been undertaken in Australia. This component of this study is aimed at determining the role of environmental flows in sediment smothering and improving/maintaining substrate sediment habitat.

6.4 Methods

Sites

The physical habitat study undertakes detailed investigations at two study sites in the lowland reaches of the Goulburn River: Darcy Track, 7km south of Mooroopna, and McCoys Bridge, to the northwest of Shepparton, on the Murray-Valley Highway. These sites were previously identified and surveyed by Water Technology Pty Ltd (Water Technology 2009) for the GBCMA as part of the Victorian Environmental Flows Monitoring and Assessment Program (VEFMAP), initiated by the Department of Sustainability and Environment (DSE). Each site covers at least one meander wavelength (down-valley distance of one complete meander) and extends for approximately 750 m. At each site the VEFMAP data contains 15 equally spaced cross-sections.

Field site inspections

Data were collected from Darcy Track during six field inspections, and from McCoys Bridge during five field inspections, from 3 October 2012 to 16 May 2013. Field inspections involved either sediment sampling, measurement of hydraulics, or a combination of the two (Figure 6.1).

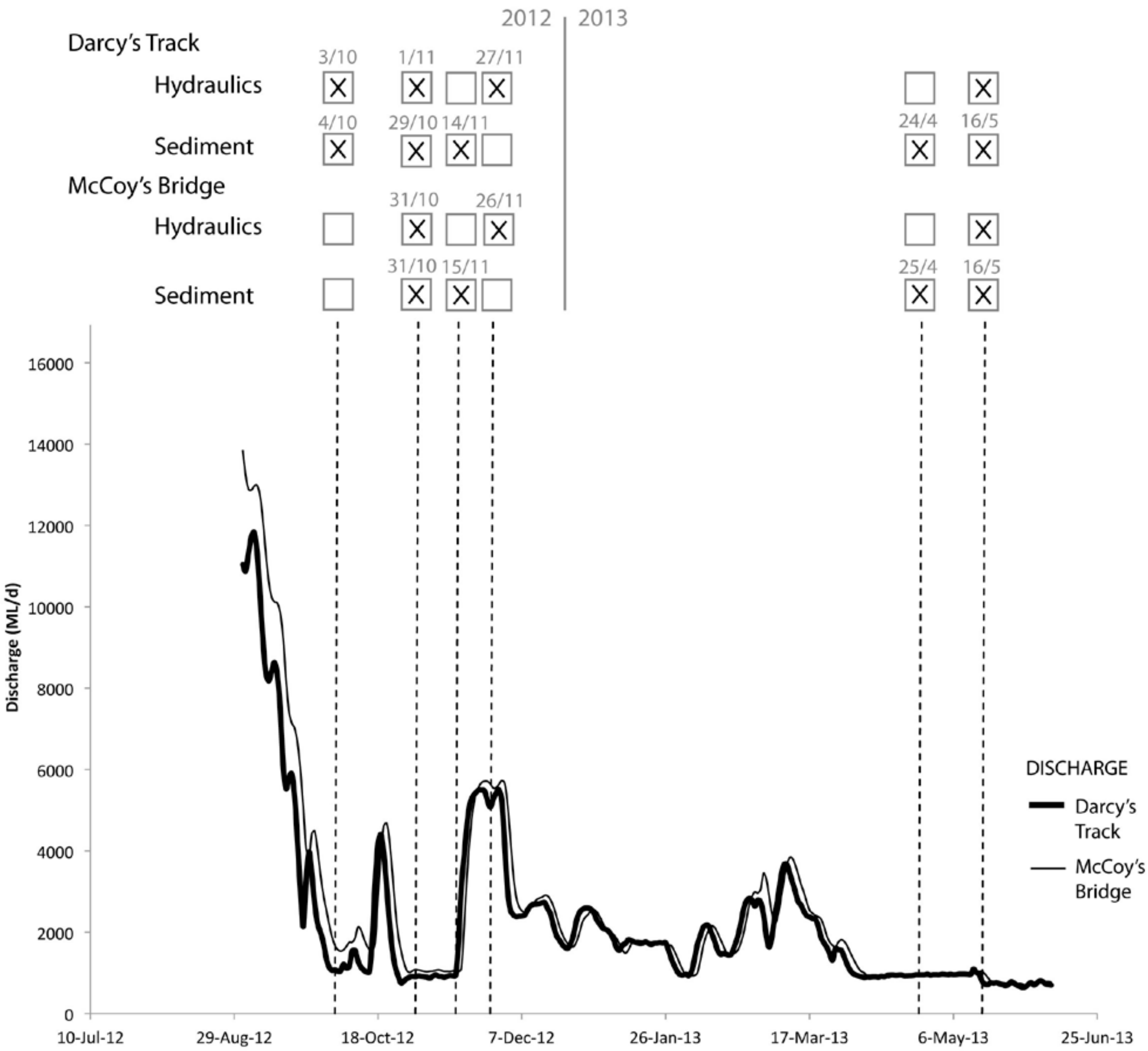


Figure 6.1: Field inspections (hydraulics and sediment sampling) relative to discharge for both the Darcy Track and McCoys Bridge sites.

Hydrology

Hydrologic data was sourced from the Victorian Water Data Warehouse (VWDW, vicwaterdatawarehouse.com.au). Goulburn River flow gauges used were McCoys Bridge (405232), and for the Darcy Track site flows were derived from the gauge at Shepparton (405204) minus inflows from the Broken River (using Broken River at Orvale, 404222). Recent flows for Shepparton (beyond 8 April 2013) - unavailable at the time of analysis - were obtained from instantaneous daily measurements supplied by Geoff Earl (Goulburn Broken CMA).

In the absence of continuous suspended sediment data, turbidity data provides a useful surrogate. Instantaneous turbidity data was downloaded for Goulburn Weir (from VWDW) and aggregated to daily values. Turbidity data was compared to the nearest available discharge data at Murchison (405200).

In order to disaggregate natural flows from environmental flows, the environmental flow release discharge data were obtained from Goulburn Murray Water (via Ross Thompson, University of Canberra).

Sediment sampling

The bed substrate at both sites was coarse-grained, mostly sand. So the determination of smothering was based on the changes in fine-grained sediment mass (silt/clay). Sediment sampling areas corresponded with suitable VEFMAP cross-sections: where initial inspection revealed coarse-grained sediments on the bed (within which fine-grained sediment could be sampled). Sites were excluded if the main bed sediment was in-situ clay. This resulted in sampling at between six and eight cross-sections at both Darcy Track (Figure 6.2), and six cross-sections at McCoys Bridge (Figure 6.3). Variation in the use of cross-sections was influenced by the movement of large wood into or out of the sampling area. Each sediment sampling area included between two and four points. A total of 170 sediment samples were collected over the study period, for subsequent laboratory analysis.

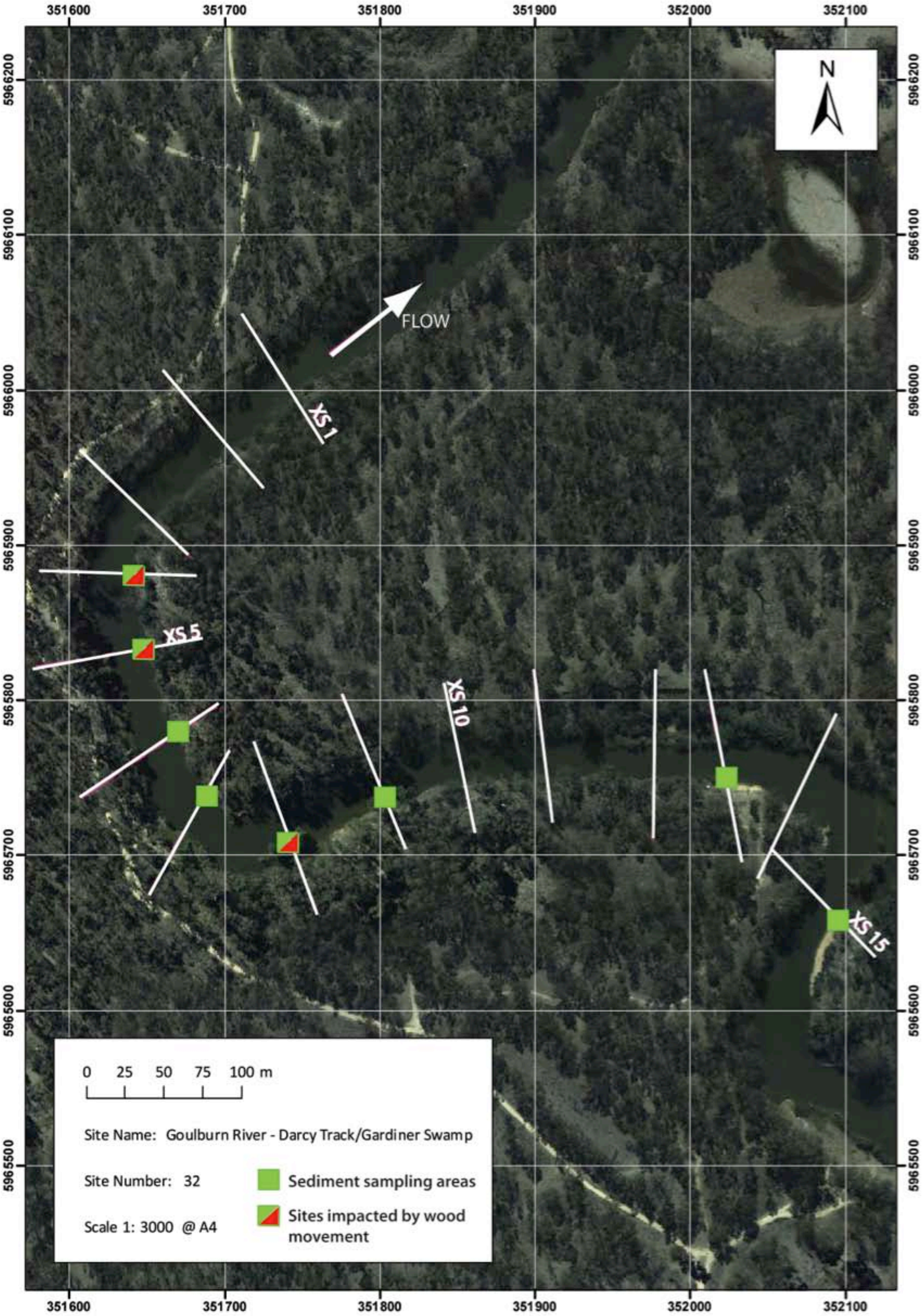


Figure 6.2: Cross sections (white lines) for Darcy Track with sampling areas indicated (green boxes) and those impacted by wood movement (green/red boxes). Modified from Water Technology (2009).

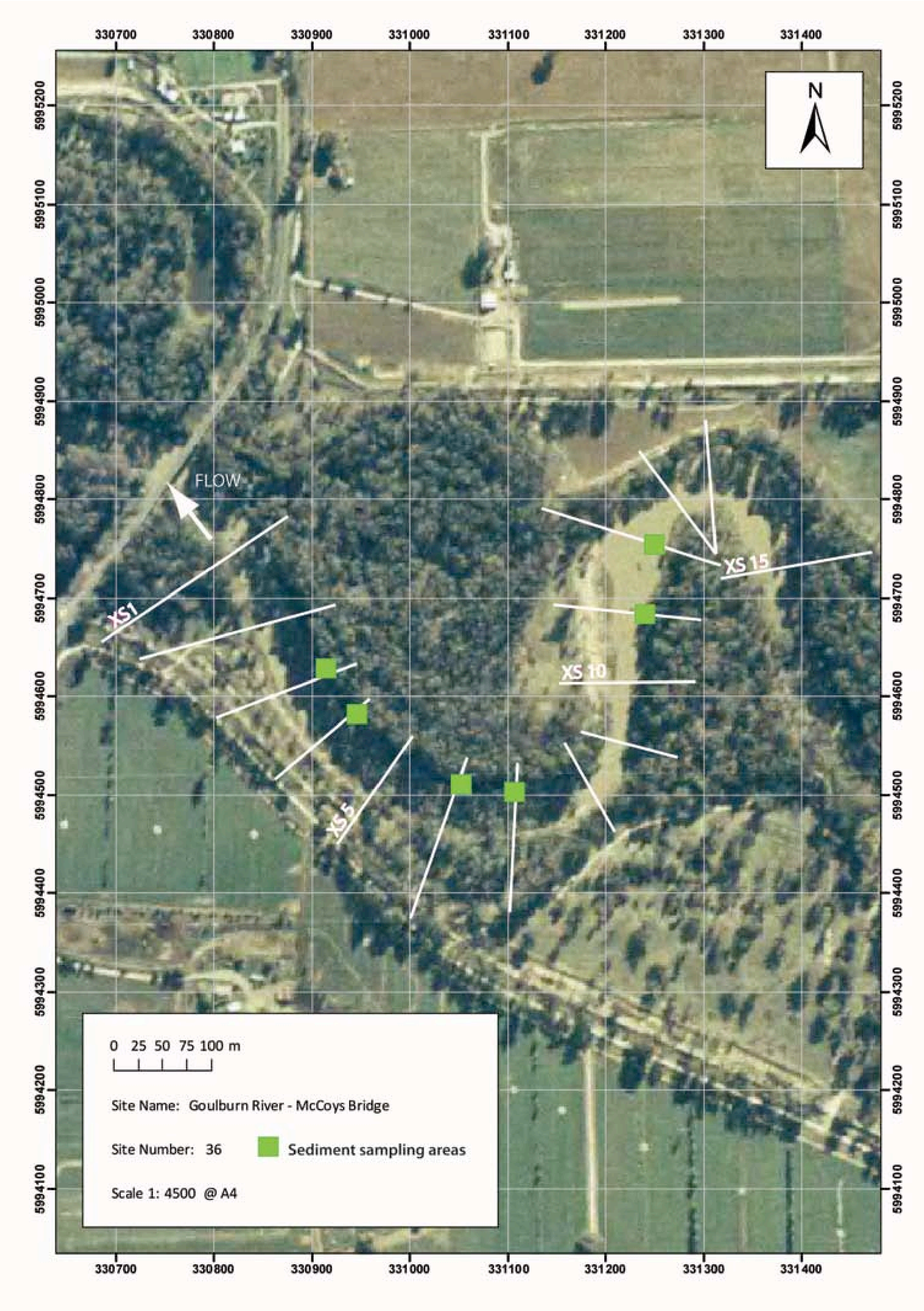


Figure 6.3: Cross sections (white lines) for McCoys Bridge with sampling areas indicated (green boxes). Modified from Water Technology (2009).

Underwater samples of fine-grained sediments cannot be simply collected by grab samples due to escape from sampling devices, so we used the re-suspension technique of Lambert and Walling (1988). Sample points for each cross-section were selected between the water's edge and the point at which the water depth reached 1 m. The 1 m depth was used as this was the depth beyond which sampling became infeasible due to the need to disturb the bed. Along a transect between these two points a random number generator was used to randomly select sampling points (Figure 6.4a).

At each sampling point a 300 mm PVC cylinder was carefully inserted into the substrate sands, and the water depth measured to isolate a known water quantity contained in the cylinder (Figure 6.4b). Using a metal rod the riverbed within the cylinder was stirred for 20 seconds to a depth of 50 mm to liberate fine-grained sediments (Figure 6.4c). Two water samples were then taken from the middle of the cylinder column using a 500 mL wide-mouthed container and combined to provide 1L sample (Figure 6.4d). Repeat samples during subsequent visits were taken at the same distance from the bank but at a slightly altered location upstream (approximately 0.5 m) to avoid resampling previously disturbed areas.

Background samples from the river were also taken. This allowed the component of the fine sediment in suspension to be subtracted from the sediment that was liberated from the riverbed. Following the first two field campaigns, and subsequent analysis, the number of background samples collected was reduced from one for each section down to two per site as the variation in samples was small as a proportion of the mean sediment mass (between 0.9 and 1.5%).

Figure 6.4: The method for field sampling of fine-grained sediment in the bed: a) sampling area measurement from bank for random sampling site selection, b) disturbance of bed sediments within cylinder, c) suspended sample collection, and d) collected sample to be transferred to combined 1L sample for subsequent analysis.

Table 6.1. Potential limitations to sampling design, and subsequent methodological considerations.

Limitation	Consideration
Limitation of maximum depths for use with the cylinder, preventing sampling in deeper water.	Nevertheless, shallow, slow flowing areas were the focus of changes in sediment smothering as they are more likely to experience low velocities and as such are more susceptible to increased smothering (Einstein 1968; Frostick et al. 1984).
Assumption that the samples from the cylinder were from a uniform sediment concentrations.	Two samples taken from the middle of the column were aimed at reducing this error and an assessment of potential error associated with sample retrieval was undertaken. For samples collected at time 0 seconds, 20, 40 and 60 seconds, analysed for mass, the standard error associated with time was negligible (0.086 kg/m^2).
Sites with high root mass, or large wood, prevented sealing of the cylinder.	Sites with high wood loads or roots covering the channel bed were not sampled and some sites had to be abandoned due to wood shifting into sampling sites.
Resampling locations had to be slightly altered to avoid areas previously disturbed.	Repeat samples were moved approximately 0.5 m upstream (upflow) of the previous sample point to avoid resampling a previously disturbed site. Results indicating increases in sediment mass for repeat samples suggest that resampling disturbed sites has not been significant.
Variability in the depth of agitation of the riverbed.	Attempts were made to standardise depths by placing rod on the channel bed and subsequently lowering by 5 cm.

Laboratory Analysis of 'Smothering' Sediment Samples

Each sample was subsequently assessed for dry sediment mass in the laboratory. The volume of each sample (including background samples) was measured prior to drying in pre-weighed aluminium trays in an oven at 80°C for at least 48 hours. The dry sediment mass, with background sediment mass subtracted, was used to determine the total mass of fine-grained sediments stored in the bed per square metre (by considering cylinder dimensions).

Small quantities of dried samples were also analysed for particle size to determine the calibre of sediment smothering the bed and substrate interstices. Samples were first mixed with a dispersant solution (10% sodium hexametaphosphate, Calgon) and passed through a 2 mm sieve and analysed using Particle Size Analysis (PSA) and a Laser Particle Sizer (Beckman Coulter LP 13320 single-wave length sizer), with particle size distributions obtained for each site (Figure 6.5).

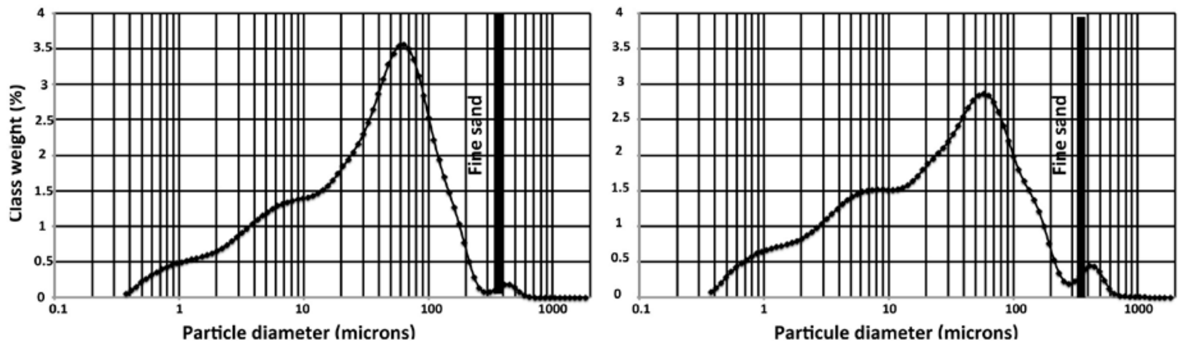


Figure 6.5: Particle size distribution for 'smothering' sediment a) Darcy Track and b) McCoys bridge for samples collected on 14-15 November 2012. The vertical line represents the upper limit particle size of fine-grained sand highlighting that smothering sediment is distinctly finer than bedload sediments.

Hydraulics

An Acoustic Doppler Current Profiler (ADCP) was used to measure hydraulic characteristics, namely discharge, bathymetry, mean velocity and depth, at both field sites. The ADCP is also used at transects to capture and verify hydraulic conditions for the use in hydraulic modeling, with a particular focus on cross sections where sediment assessments were undertaken.

The ADCP is mounted onto a custom-made board dragged across the river either by rope (at low flows, approx. 1000 ML/d) or canoe (at high flows, approx. 5000 ML/d) (Figure 6.6). This is because the width of the channel, and flow velocity, meant that the more accurate rope method was not feasible at high flows. At least two passes of the ADCP were undertaken and the values averaged. ADCP data were analysed using Matlab to produce data for verification and calibration of the one-dimensional hydraulic models (HEC RAS).

Figure 6.6: The use of the ADCP to assess field hydraulics along transects using a) roped method, and b) trailed behind a canoe.

The one-dimensional hydraulic modeling tool (HEC RAS) was used by Water Technology (2009) to develop models for each site, based on 2009 survey topography. To determine changes in bed morphology since the Water Technology bed survey we verified against bed profiles determined by the ADCP and found general agreement for most bed profile patterns, shown here for high flows (Figure 6.7a and b). An exception to this is cross section 10 at Darcy where the ADCP data displays a considerably narrower channel than for the survey data (Figure 6.7a), an anomaly that we suggest is due to an error in the ADCP bottom tracking rather than survey error or channel change, and one that would only have a minor influence on the proportion of slackwater habitat as determined by the ADCP. For McCoys while the data would suggest degradation (erosion) of the channel, the differences are most likely due to fitting of the data using discharge as the common datum (Figure 6.7b). Morphologic patterns display little change.

Whilst the ADCP was intended to be used to quantify both deepwater and shallow water habitat it was found to be prone to large errors in velocity readings for shallow environments (<0.5 m). For this reason a handheld Acoustic Doppler Velocimeter (ADV) was also trialed toward the end of the field campaign (16/5/2013). Whilst only providing one point in time the results of the ADV were used to assess velocities relative to sediment smothering.

Development of physical habitat models

Slackwaters were defined by low velocities less than 0.05 m/s, and shallow depths less than 0.5 m. The first quantification of changes in slackwater habitat with discharge was determined for the nearby Broken River and recently published (Vietz et al., 2013). This relationship quantifies slackwater habitat based on relative discharge (relative to bankfull discharge) for use internationally in morphologically similar channels to that of the Broken River. Considering the Broken River is a tributary of the Goulburn, within similar geomorphology, and they both exhibit bar and bench morphology, the relationship was deemed suitable for application to the Goulburn River. We also compared slackwater abundance determined from ADCP data and, whilst only for three discharges, found a similar trend with respect to discharge to that represented by the generic curve.

Comparison of slackwater area determined from the field sampled ADCP data for two sample periods (low flow and fresh) suggests that either the relationship developed for the Goulburn River sites underestimates the amount of slackwater habitat, or the ADCP data (two points) overestimates slackwater habitat. Nevertheless, comparison of the two demonstrates good relative agreement. For higher flow there is decreased slackwater habitat in both cases. Since the relative difference in slackwater availability is of interest, and in light of the lack of a two-dimensional model for the Goulburn River, this approach has been deemed appropriate for this study.

Deepwater habitat was determined based on the one-dimensional HEC RAS hydraulic model output. The availability of deepwater habitat (defined as depth > 1m) was determined for every one of the 15 cross sections, for 21 discharges.

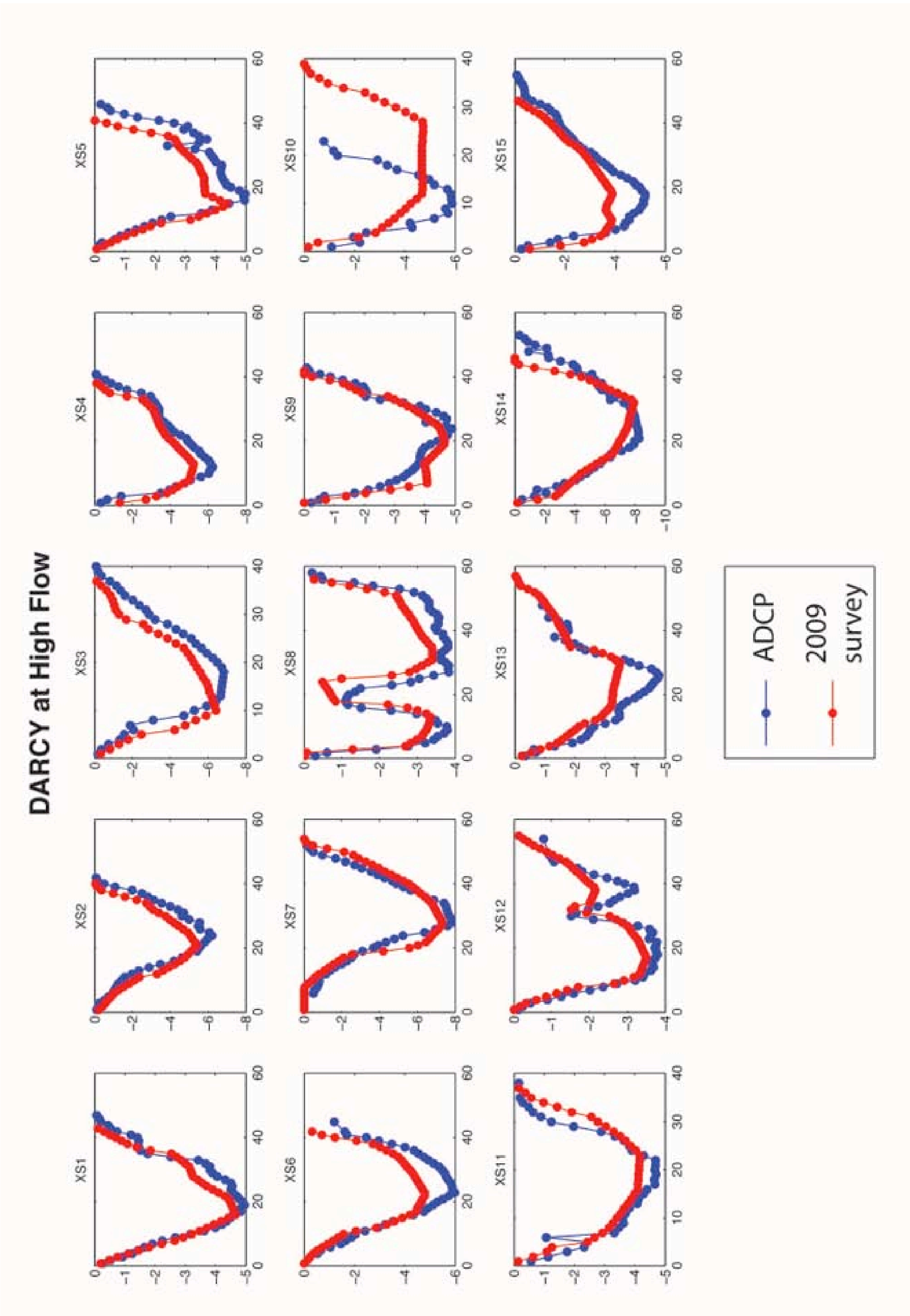


Figure 6.7a: Comparison of ADCP and 2009 survey bed profiles (Darcy Track, High flow) to ensure applicability of using field measurements to verify hydraulic modeling.

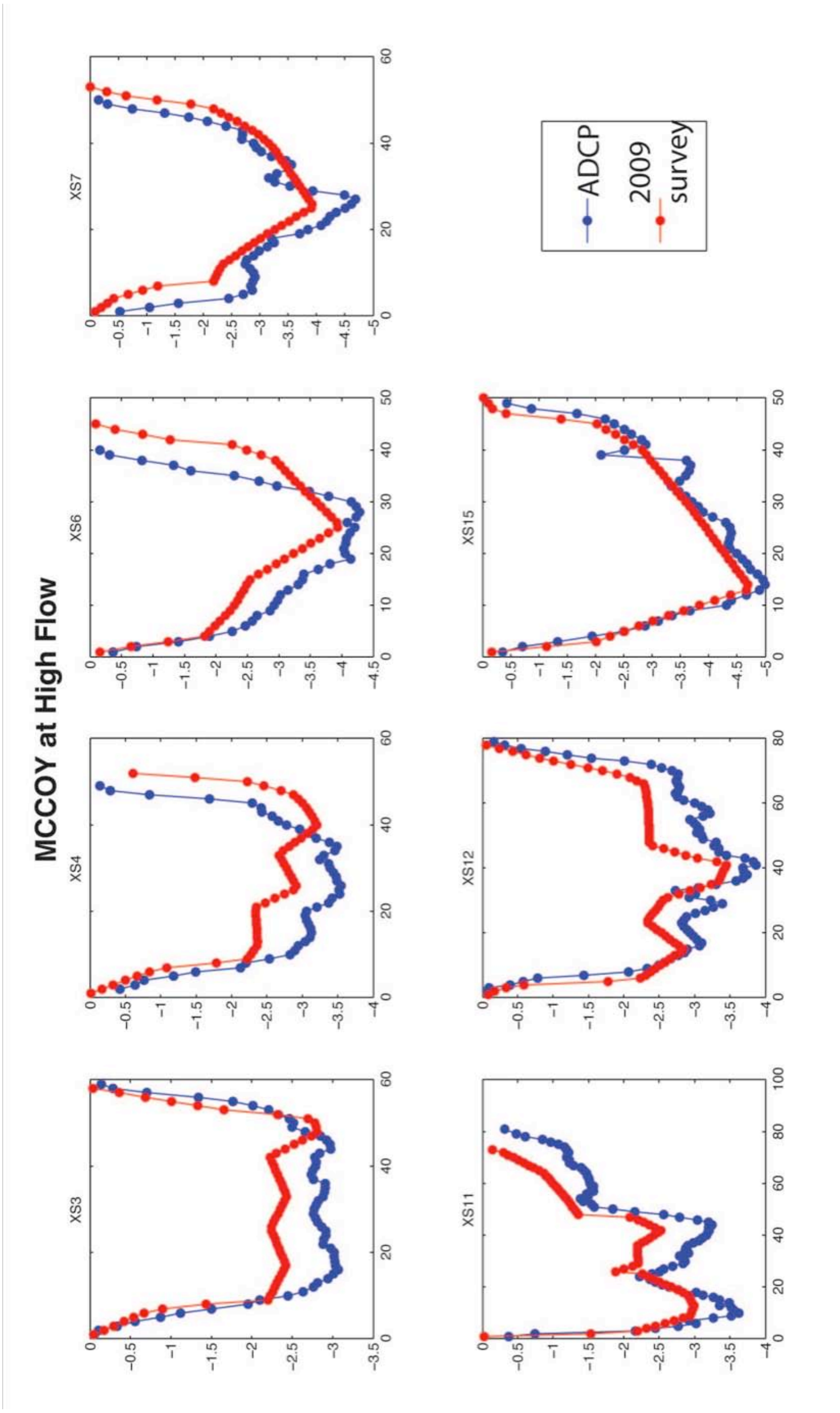


Figure 6.7b: Comparison of ADCP and 2009 survey bed profiles (McCoys Bridge, High flow) to ensure applicability of using field measurements to verify hydraulic modeling.

6.5 Results

Slackwater habitat

Changes in channel discharge alter both velocity and depth distributions. Increases in discharge initially increase slackwater habitat for very low flows (as the bed is inundated) and as discharges increase to fill the channel both depth and velocity increases with decreasing slackwater area until bars and benches are inundated. This relationship results in baseflows increasing slackwater habitat by elevating discharge above otherwise very low flow conditions, and freshes decreasing slackwater habitat by elevating discharge to levels where depth and velocity exceed that which defines slackwaters.

Based on the relationship for discharge and slackwaters, and comparing the hydrographs with and without environmental flows, provides some trends on the influence of slackwaters. For the study period the number of days of slackwater habitat over 100 m^2/100 m is increased overall with environmental flows: Darcy Track increased from 81 to 101 days, and McCoys Bridge from 76 to 94 days (for a total of 176 days over the study period), Figure 6.8. For McCoys Bridge the total area of slackwater habitat also increased over the study period (+4%), however, for Darcy Track there is a decrease in total area of slackwater habitat (-10%). Overall, slackwater habitat area is considerably greater for Darcy Track than for McCoys Bridge.

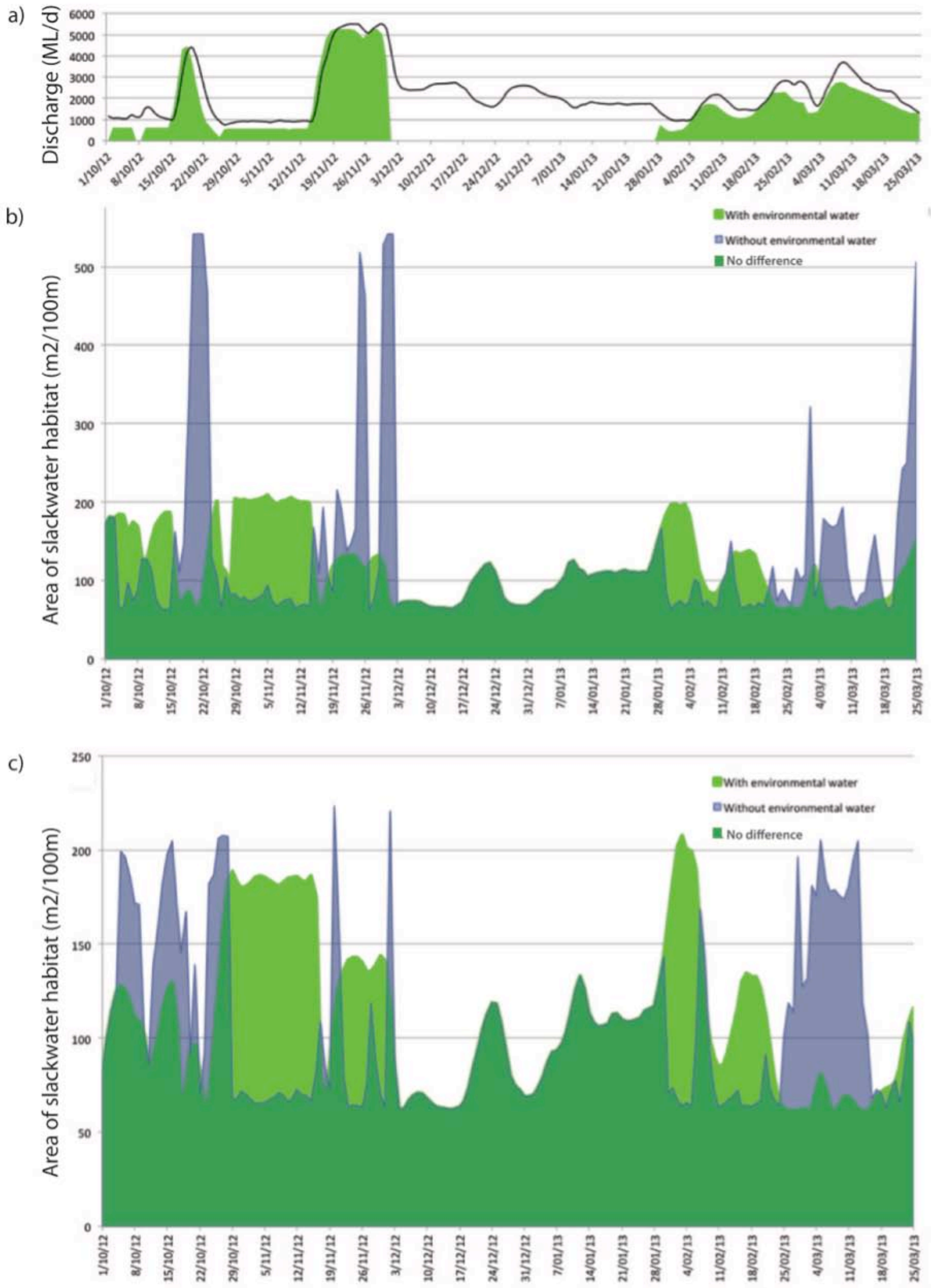


Figure 6.8: Comparison of slackwater habitat availability with environmental flows (green) and without (blue) for Darcy Track (b) and McCoys Bridge (c). The flow hydrograph (a) is for Darcy Track.

Deepwater habitat

Deepwater habitat, defined as the area of pools with depth > 1m, increases rapidly for lower discharges (<2000 ML/d), and for greater discharges responds almost linearly for both Darcy Track (Figure 6.9) and McCoys Bridge (Figure 6.10). Base flows (900-1000 ML/d) provide considerable increases in deepwater habitat of more than 50% when compared with low flows (100-200 ML/d). Flow freshes (5000-6000 ML/d) also provide increases in deepwater habitat, although the gains in habitat relative to discharge diminish for flows greater than 2000 ML/d.

Over the irrigation season environmental flow delivery increases the area of deepwater habitat by 32% and 34% for Darcy Track (Figure 6.11) and McCoys Bridge (Figure 6.12.), respectively (October to March inclusive). The number of days that deepwater habitat was provided also increased over the study period. For 150 m^2/100 m of deepwater habitat environmental flows increase the number of days from 47 to 174, and 107 to 176, for Darcy Track and McCoys Bridge, respectively. Overall, the McCoys Bridge site provides a greater area of pool habitat.

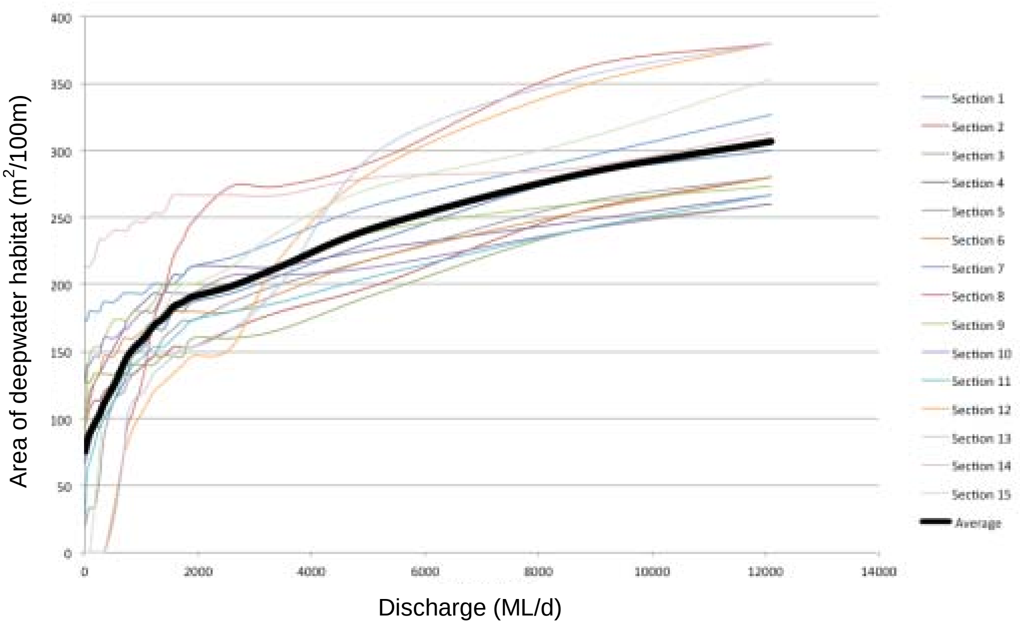


Figure 6.9: Change in the area of deepwater habitat (area of depth > 1.0m per 100m of stream) against discharge to bankfull for Darcy Track. The general trend is the average (black line) of all 15 sections (coloured lines).

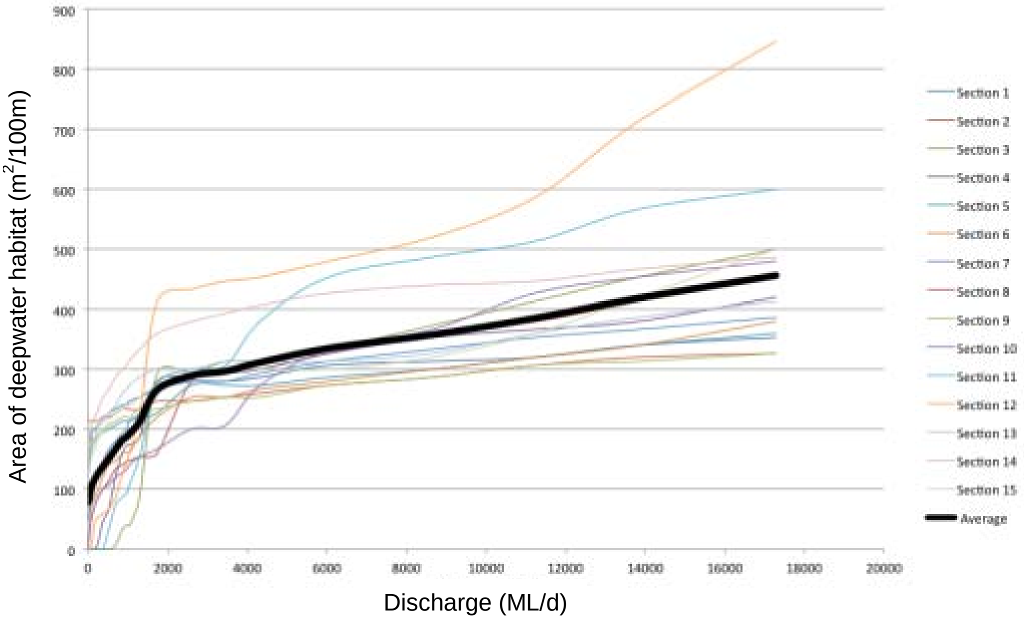


Figure 6.10: Change in the area of deepwater habitat (area of depth > 1.0m per 100m of stream) against discharge to bankfull for McCoys Bridge. The general trend is the average (black line) of all 15 sections (coloured lines).

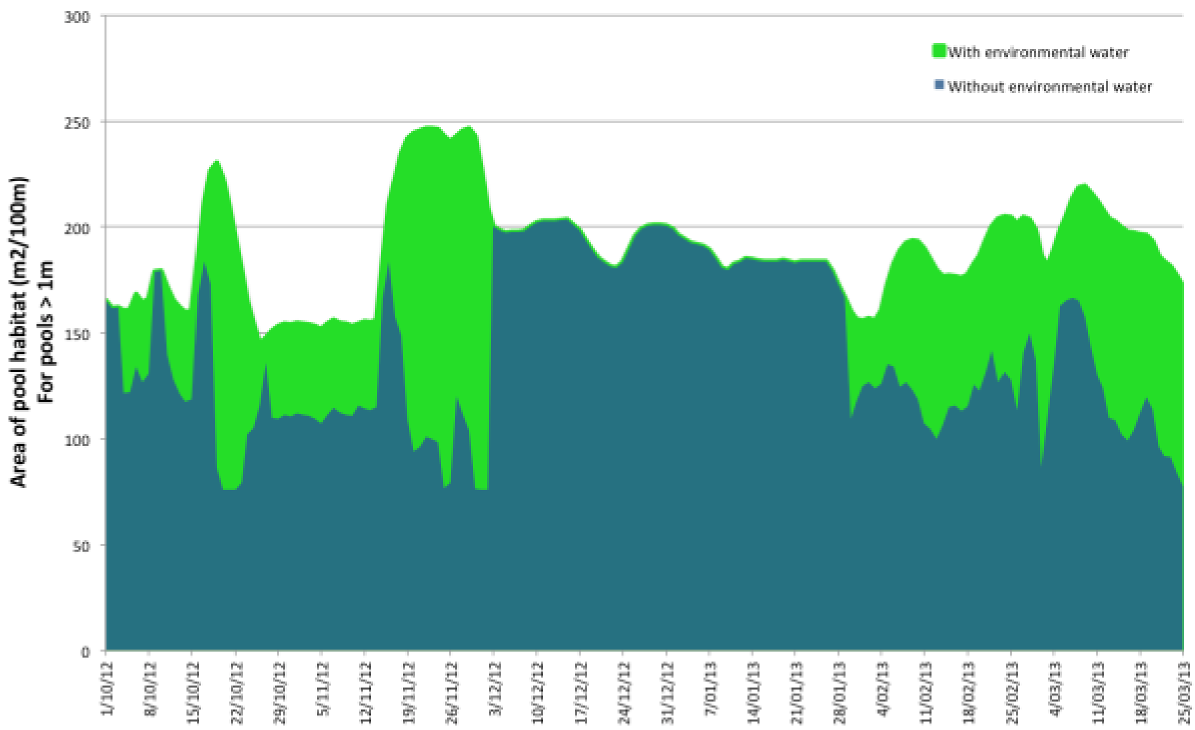


Figure 6.11: Area of pool habitat (m^2/100 m of stream reach) for Darcy Track for the flow regime with environmental flows (green) and without (blue).

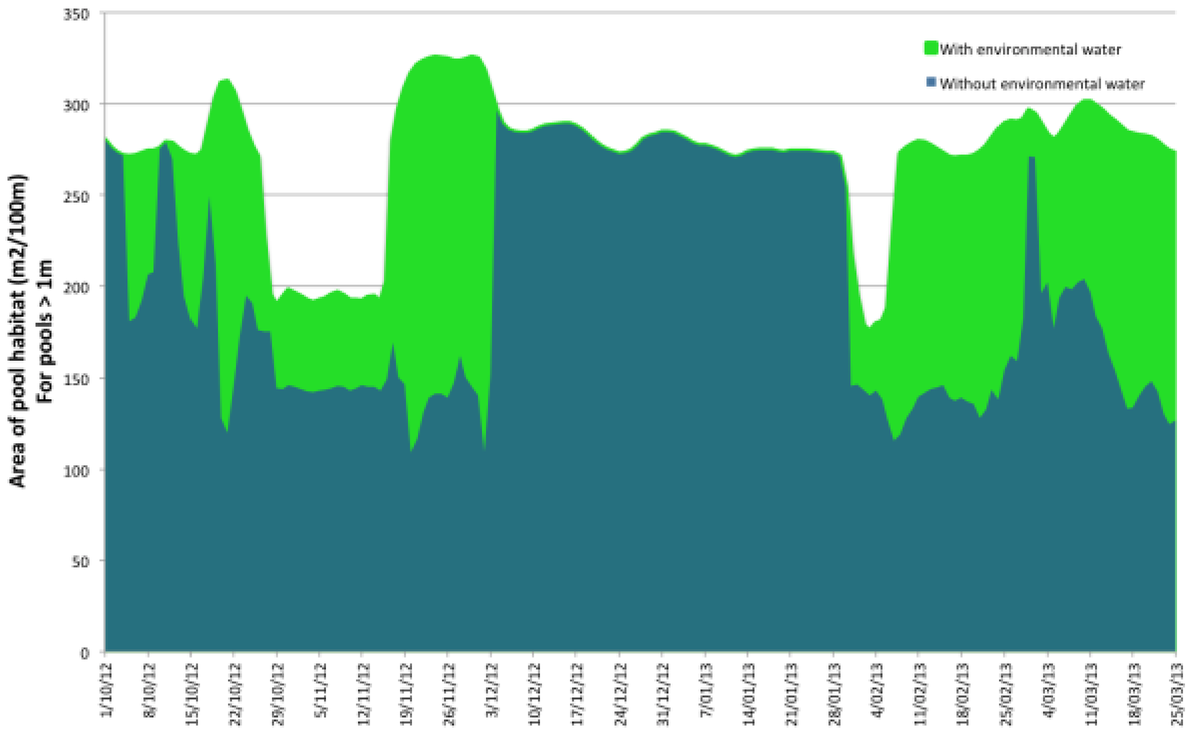
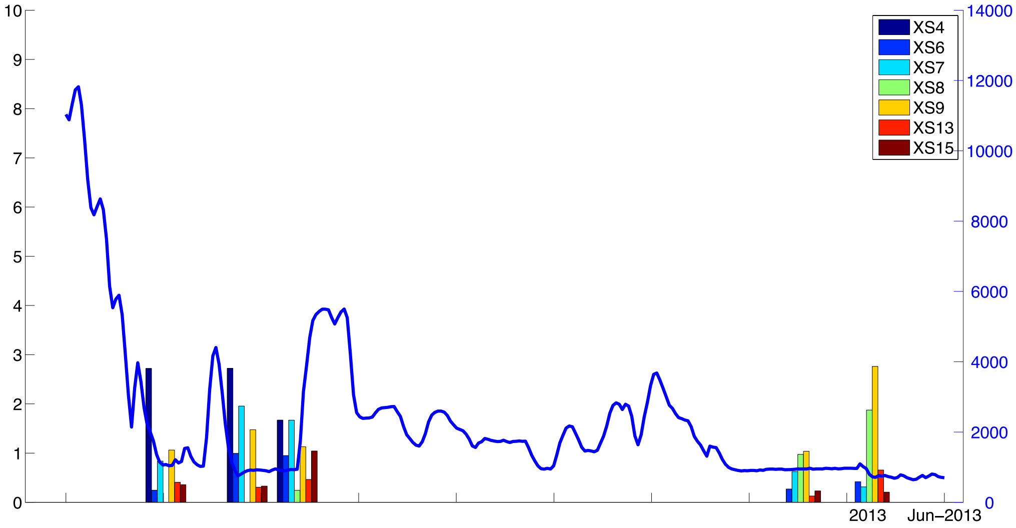


Figure 6.12: Area of pool habitat (m2/100 m of stream reach) for McCoys Bridge for the flow regime with environmental flows (green) and without (blue).

The impacts of environmental flows on sediment smothering

We hypothesised that sediment smothering would increase during low flow periods and decrease following flow freshes. We found that sediment smothering is in fact highly variable spatially both between sample points (within cross-sections) and sample areas (from one cross-section to another), at both sites investigated. Nevertheless, by aggregating sample points at each transect and presenting results by mass per sample area (kg/m^2) some trends relative to discharge are evident (Figure 6.13).

Variations with time result in increases in smothering at one site, and decreases in another, for the same sample period (e.g. Section 7 and 9 Apr and May, Figure 6.13). Some of this variation can be associated with large wood. Sections of the river channel with large wood (at the meander bend scale) are less impacted by variations in velocity, providing a roughness element directing high velocity away from the site thus reducing velocities.



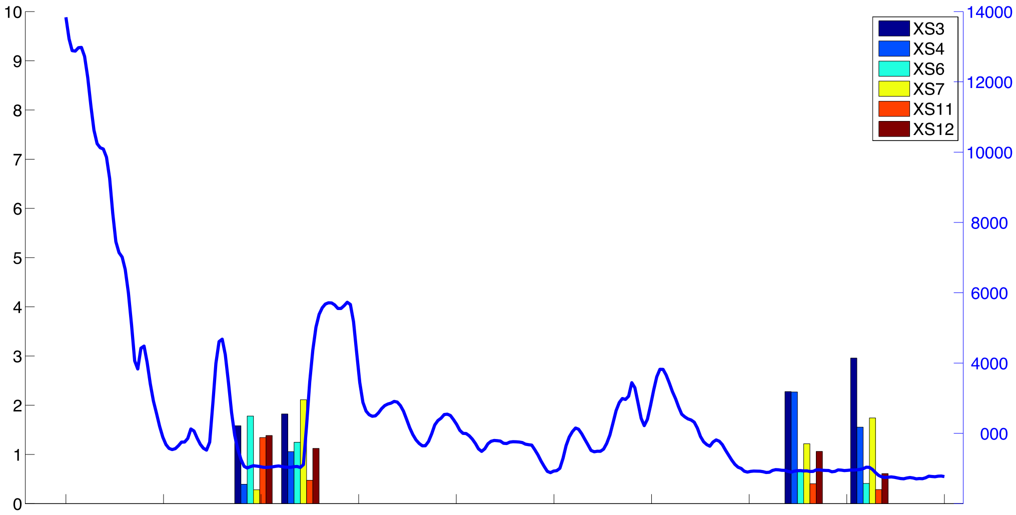


Figure 6.13: Discharge relative to the mean mass of sediment smothering the bed (kg/m^2) for (a) Darcy Track and (b) McCoys Bridge. Mass is presented for each cross section (cumulative result for the three samples) for each sampling campaign.

The median mass, and percentile range, of settled sediments for each sampling campaign demonstrate few significant relationships. For Darcy Track (Figure 6.14a) there is an initially low mass following the large and prolonged September flow event. The paired sampling campaigns following the late October environmental flow fresh display no significant difference, and there is also no significant difference following the numerous flow freshes to April. The results are similar for McCoys Bridge (Figure 6.14b). The only difference that could be drawn is for lower sediment masses following the numerous flow freshes to April compared with that for the sampling prior to these events (end of November compared to early May, respectively) at Darcy Track.

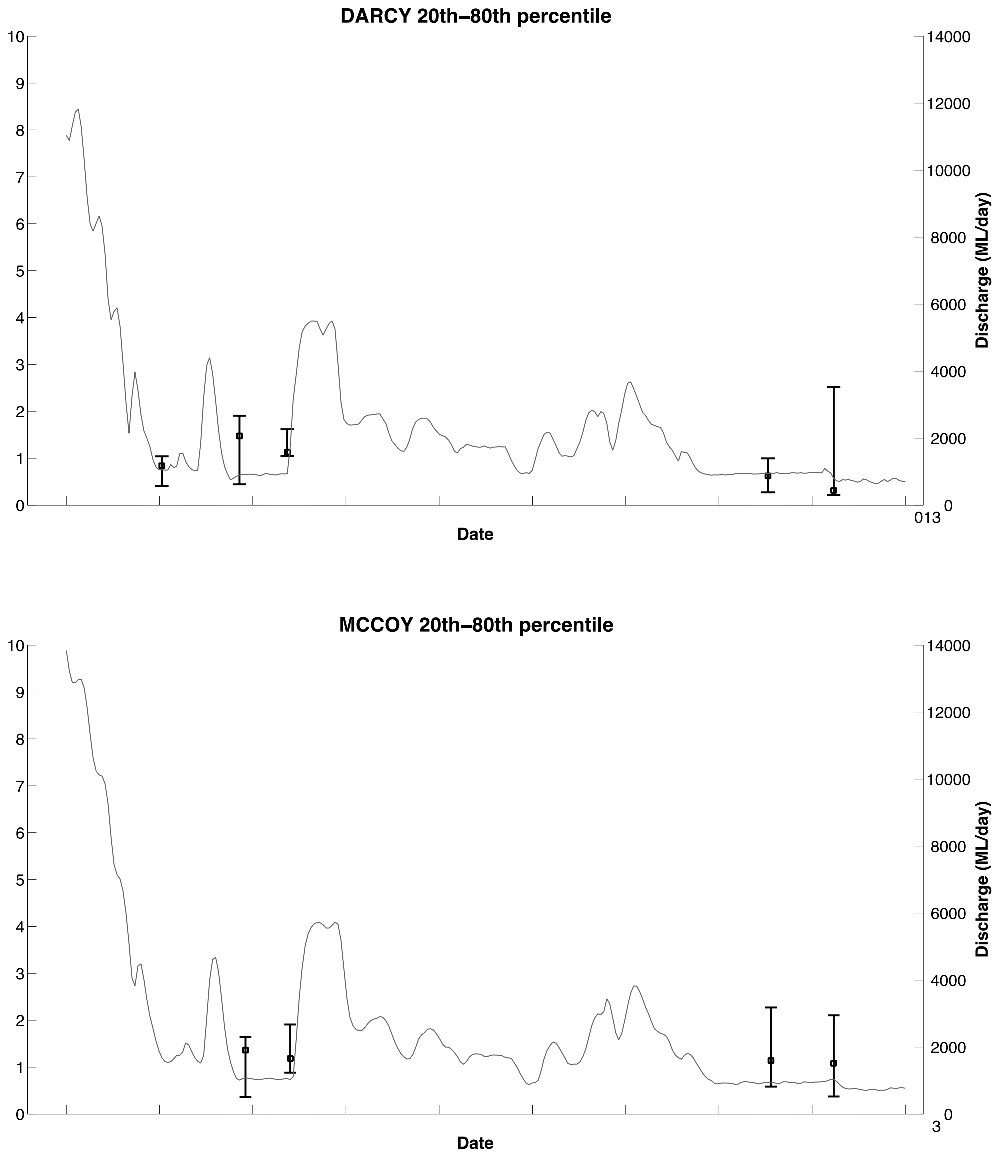


Figure 6.14: Discharge relative to the median mass of all sediment samples ((kg/m^2, and 20th and 80th percentiles) for (a) Darcy Track and (b) McCoys Bridge. Mass is presented as cumulative result for all site samples from each sampling campaign.

The sediment that smothers

The sediment that smothers the bed, as measured by this method, can be classified as mud: silts and clays (68%), with some fine to very fine sand (approx. 30%), Table 3-13. This sediment is distinct from the medium/coarse sands, and gravels that commonly comprise the bed substrate of the Goulburn River (Water Technology 2009), although a small proportion of re-suspended sediment contains sand sized particles.

Table 6.2: Particle size distribution, based on *Wentworth (1922).*

	Descriptive Term	Size	MCCOY		DARCY	
% GRAVEL:	V COARSE GRAVEL	mm 32	0.00%	0.00%	0.00%	0.00%
	COARSE GRAVEL	16		0.00%		0.00%
	MEDIUM GRAVEL	8		0.00%		0.00%
	FINE GRAVEL	4		0.00%		0.00%
	V FINE GRAVEL	2		0.00%		0.00%
% SAND:	V COARSE SAND	1	28.44%	0.02%	31.76%	0.00%
	COARSE SAND	microns 500		0.78%		0.24%
	MEDIUM SAND	250		2.37%		1.01%
	FINE SAND	125		7.79%		8.33%
	V FINE SAND	63		17.48%		22.17%
% MUD:	V COARSE SILT	31	71.56%	19.23%	68.24%	22.07%
	COARSE SILT	16		14.03%		13.92%
	MEDIUM SILT	8		11.35%		10.51%
	FINE SILT	4		10.46%		8.94%
	V FINE SILT	2		7.32%		5.91%
	CLAY			9.17%		6.89%

Sediment smothering and velocity

The deposition and scouring of fine-grained sediment can be related to average velocities over the sampling point. Based on field-collected measurements of velocity using the ADV (found to be more reliable in shallow water depths than the ADCP) we use velocity as a surrogate for shear stress. A plot against sediment smothering identifies a weak negative trend between velocity and the mass of smothering sediment. In general low velocities relate to a complete range of sediment smothering, whereas high velocities lead to low levels of sediment smothering. A large proportion of the variability could be attributed to the velocity measurement being a point in time, whereas, the level of sediment smothering is a function of the antecedent conditions. Velocity measurements over an extensive range of discharges or a two-dimensional hydraulic model would enable a more integrative approach to the role of antecedent velocity. Most importantly an ecological study investigating acceptable levels of sediment smothering would assist in driving flow management that maintains these levels e.g. if 1 kg/m^2 was acceptable then 0.25 m/s would achieve acceptable conditions.

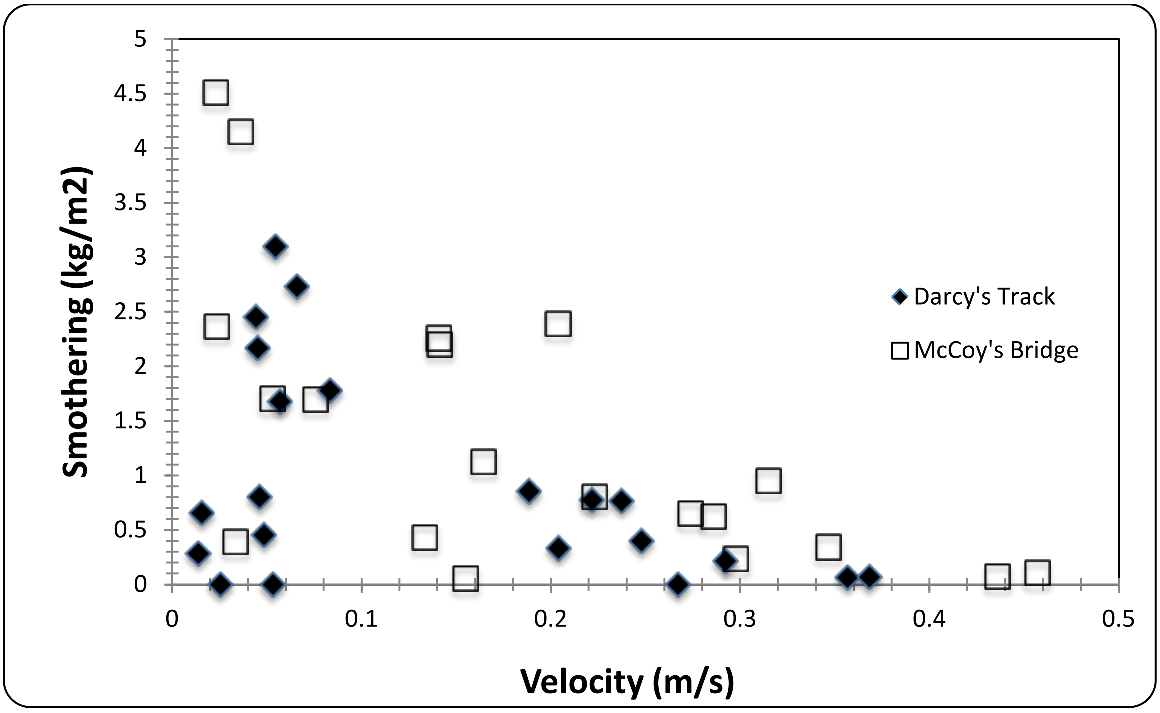


Figure 6.15: Sediment smothering (kg/m2) relative to field measured point velocities.

The role of sediment concentration

Suspended sediment concentration plays an important role in the availability of sediment for smothering. An important question is whether environmental flow releases alter the suspended sediment concentration, and hence influence smothering of the channel bed. Continuous data for turbidity provides a useful surrogate for investigation of this question in the absence of repeat suspended sediment sampling.

Comparison of discharge data - total flow and environmental flow component – for the study period (Figure 6.16) may illustrate some weak associations between flows and turbidity. Turbidity concentration appears to be more strongly associated with watershed derived (rainfall) events with spikes in turbidity associated with the two December events. A peak was evident with the first environmental flow (October), but not the second, and largest, in November. The increase in turbidity associated with the first environmental flow is; however, not necessarily due to the flow as the turbidity was already rising, illustrating a poor relationship. Trends in turbidity may be more readily explained by antecedent conditions.

There is a general decrease in turbidity over the study period due to the large flow event that occurred in spring 2012, Figure 6.17. Following the event peak of 22,000 ML/d there were considerable increases in turbidity on the receding limb. During this period anecdotal evidence suggests mass failure of banks was commonly observed by Catchment Management Authority staff (Cottingham et al. 2013). Increases in turbidity post-event occurred prior to environmental flow deliveries, presumably caused by drying and desiccation of banks and subsequent failure. Increased bank failure and turbidity may, however, have been exacerbated by the first environmental flow event delivered (in October). Demonstrating relationships between environmental flows, bank notching, and bank slumping following drawdown would assist in understanding how they may contribute to increased suspended sediment concentrations.

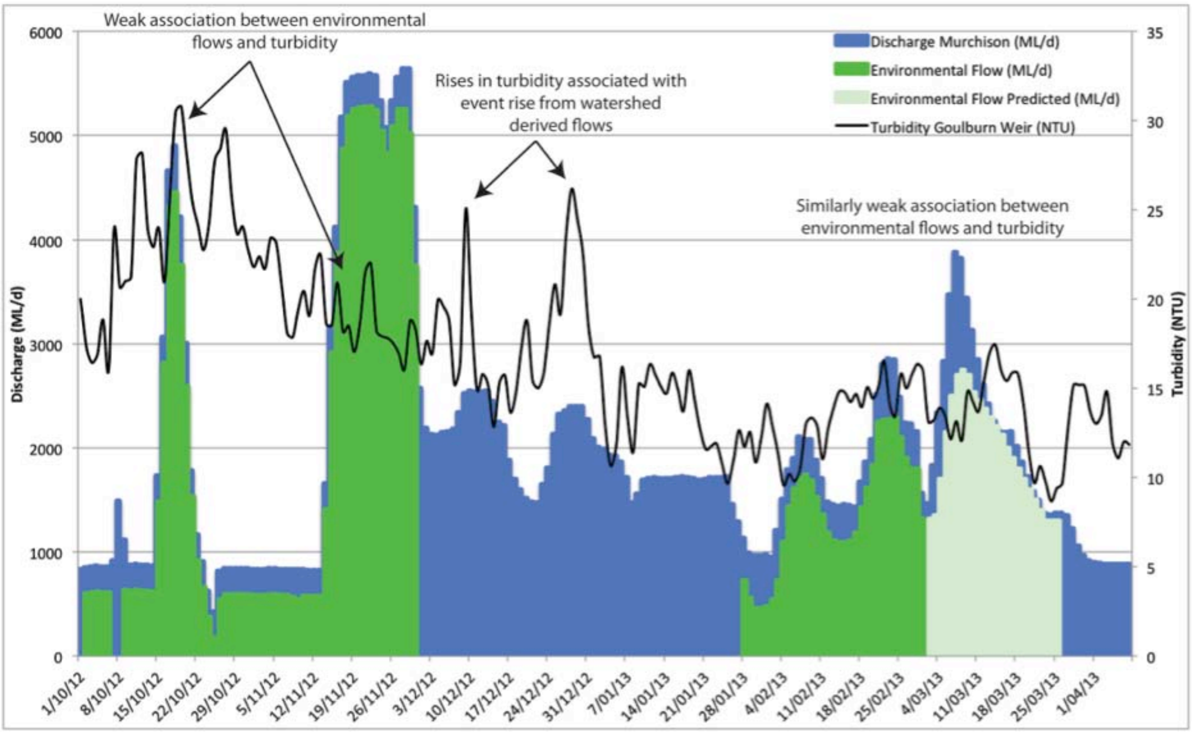


Figure 6.16: Turbidity plotted against discharge for the period leading up to and including the study period (turbidity for Goulburn Weir and discharge for Murchison).

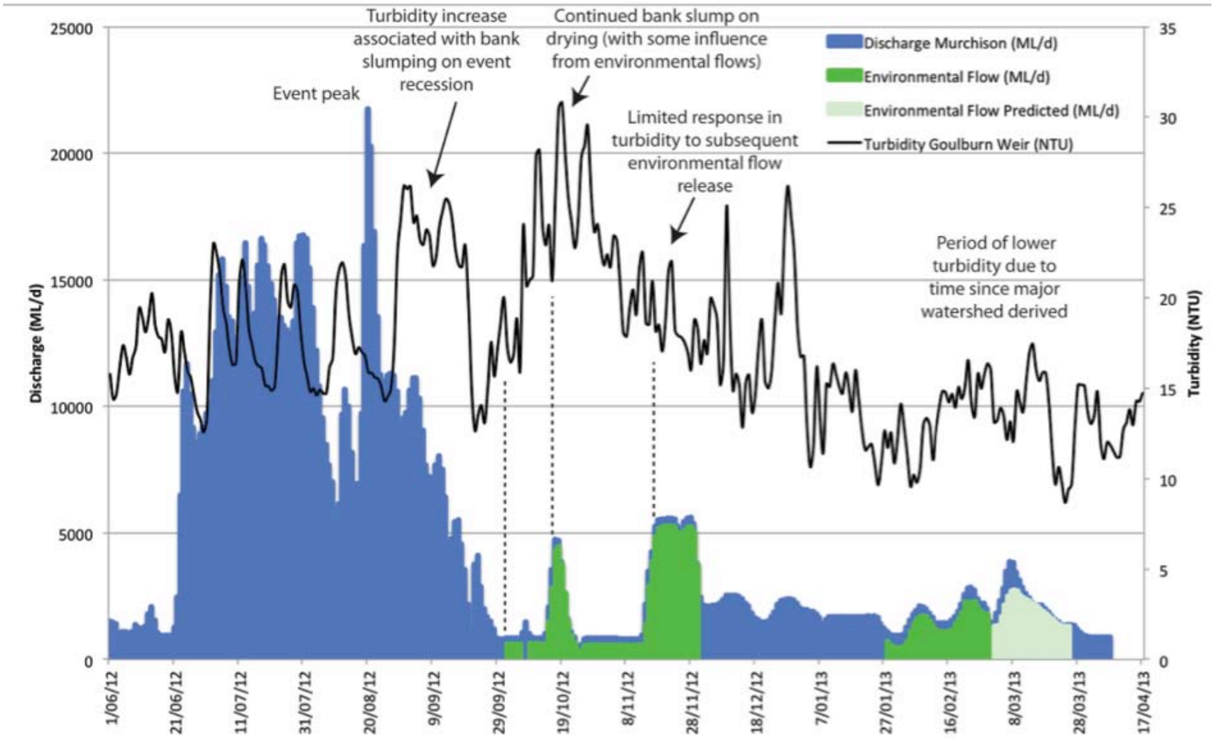


Figure 6.17: Turbidity plotted against discharge for the study period highlighting that turbidity is more influenced by watershed derived flow events than environmental flows.

6.6 Discussion

Physical habitats - specifically the changes in depth, velocity and sediment - are dependent on discharge and channel morphology. Discharge is an important driver of physical habitat and therefore modifications to the flow regime - including freshes, bankfull events and baseflows - alter the abundance and distribution of physical habitat. This research has provided the initial basis for quantification of the benefits of environmental flow delivery relative to deepwater and slackwaters, whereas the influence of environmental flows on sediment smothering is inconclusive.

Impacts of environmental flows on slackwaters

At lower discharge slackwaters often occupy large portions of the bed of the channel and for higher discharges they contract to the channel margins, as smaller less contiguous patches, Figure 6.18. Channel features such as bars and benches can provide small increases in slackwater habitat availability at higher discharges Figure 6.19. This highlights the importance of maintaining these features such as benches, thus decreasing the sensitivity of physical habitat to flow regulation. While secondary influences of environmental flows on channel morphology (and hence physical habitat) were not assessed within this study, the maintenance of bar and bench morphologies has been found to require periodic medium to high flow discharges to transport sediments to higher elevations, without prolonged high flows (Vietz et al. 2007).

Environmental water delivered as baseflows considerably increases the availability of slackwater habitat. Flow freshes, however, can decrease the availability of slackwater habitat due to increases in depths and velocities as water level moves up the banks. Nevertheless, the increase in the number of days of slackwater availability (slackwater habitat > 100 m^2/100m) for both sites suggests that environmental flows provide consistent periods of elevated slackwater availability, interspersed by decreases in peak slackwater availability during the provision of flow freshes.

Figure 6.18: Changes in slackwater habitat from being central within the channel to contracting to the channel margins for base flows (~1000 ML/d) to flow freshes (~5000 ML/d). Demonstrated for Darcy Track.

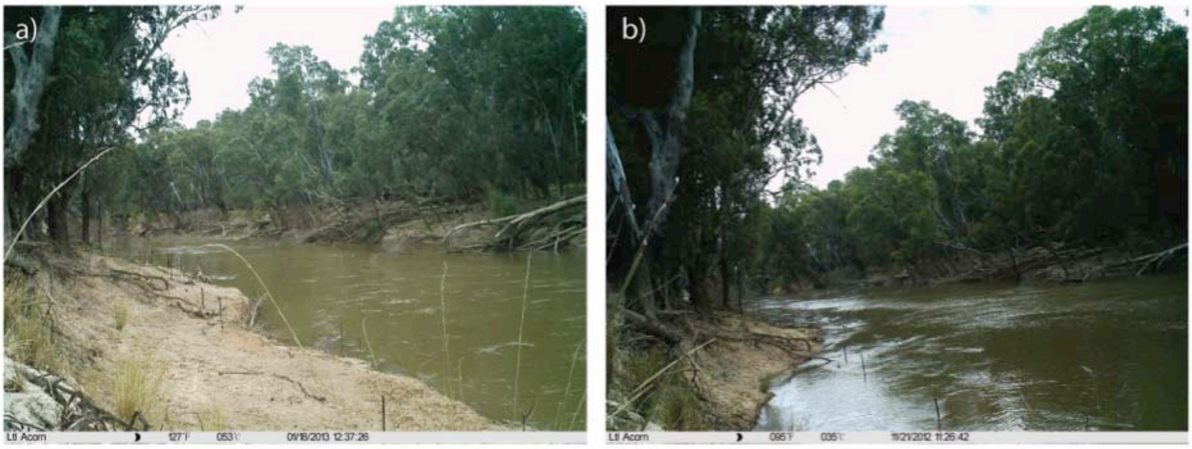


Figure 6.19: A localised increase in slackwater habitat can result from the inundation of higher level bars or benches. A bench is visible at the McCoys Bridge site (for a discharge of 1800 ML/d (a)) and is inundated at a discharge of 5300 ML/d (b)).

Impacts of environmental flows on deepwater habitat

The area of deepwater habitat increases with discharge, and so environmental flow provisions consistently lead to greater availability of deepwater habitat. Base flows provide the greatest returns in deepwater habitat with increases in flow, with diminishing returns in habitat for higher flows (greater than 2000 ML/d). The area of deepwater habitat with environmental flows is increased by more than 30% by area, and the number of days is increased by between 65 and 135% over that without the provision of environmental flows.

Water levels associated with greater deepwater habitat also lead to inundation of large wood associated with tree roots, or woody debris at the toe of bank, Figure 6.20.

Figure 6.20: Inundation of channel margins and large wood associated with water depth increases from ~1000 ML/d (a and c) to ~5000 ML/d (b and d), at Darcy Track.

Impacts of environmental flows on sediment smothering

Sediment smothering is visually evident in the shallows at the channel margins of the Goulburn River (Figure 6.21). Variability in sediment smothering of the bed from one bank to another is the result of the inherent variability in hydraulics (velocity and depth). In particular there is a weak negative correlation between velocity and sediment smothering: higher velocities lead to lower levels of smothering.

The transport of sediment that smothers the bed is likely to be episodic, such that sediment removed from one site may make its way to another and be deposited for a short time period.

The relationship is therefore not as simple as the hypothesis that low flows allow sediment to accumulate and flow freshes scour sediment from the bed. Lower flows (environmental base flows and freshes) are expected to maintain velocities that prevent sediment smothering, however, results do not show a significant trend. It may also be likely that the spatial variability in velocity distribution mean that sediment smothering is likely until large events elevate channel flows over the entire bed to levels that will induce scour. These velocity thresholds can be best determined through two-dimensional hydraulic modeling. Spatial and temporal variability in sediment smothering makes pre and post flow event differences difficult to decipher.

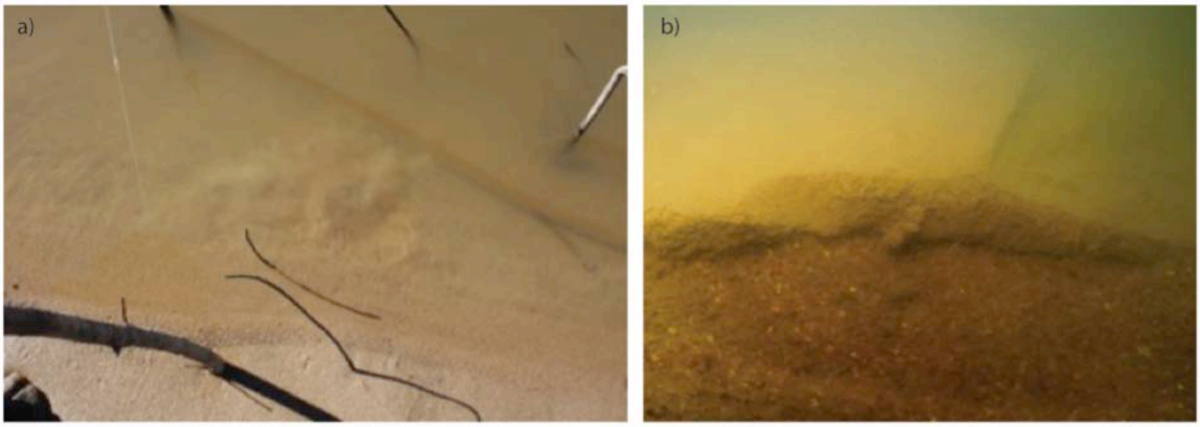


Figure 6.21: Fine-grained sediments settling and 'smothering' mobile sandy substrates at a) channel margins, and b) underwater close-up of the channel bed (with smothering partially washed away by hand), Darcy Track.

Despite the uncertainty behind sediment smothering and flows it is evident that the bed of the channel is the dominant sink of suspended sediment deposition. SEDNET modeling of the Goulburn River by Wilkinson et al. (2005) found that, 38% of incoming sediment was deposited on the bed, with 18% trapped behind weirs, 19% deposited on floodplains, and 25% is exported from the catchment. Increases in catchment derived sediments, and certainly bank erosion (Cottingham et al. 2013), may increase sediment deposited on the bed of the channel.

The release of environmental flows following large events carries a risk of bank mass failure, which may contribute to elevated sediment concentrations. Erosion of banks through mass failure is most common on the receding limb of the hydrograph, as evident from the September high flow event, and subsequent removal of the failed sediments for transport within the river (Cottingham et al. 2013). The environmental flow delivered immediately post the September high flow event may have served to remove failed sediment earlier in the irrigation season: failure that was still likely from a catchment derived event.

High turbidity associated with rainfall events (e.g. two events in December) may suggest that these have higher sediment concentrations than environmental flow releases from storage. The availability of daily suspended-sediment data (rather than merely turbidity data) would provide considerably greater potential for understanding the role of flow delivery, rainfall, and bank failure on sediment smothering.

Environmental flow events from late November through to April, including base flows and flow freshes, are anticipated to prevent increases in sediment smothering over the irrigation period, but the results from this initial short-term study are not conclusive.

The role of bedload sediments in substrate condition

The availability of coarse-grained bedload sediment may also play an important role in the condition of substrate sediment. Of the sampling areas assessed only 25% contained bedload sediments at each site, the remainder being in-situ clay (total possible sites was 30: 15 sections, 2 bank sides). This leads to the question of whether the well documented trapping efficiency of dams (Petts and Gurnell, 2005) is starving the Goulburn River system of coarse-grained sediments.

Erskine et al. (1993) calculated the trap efficiency of Eildon Dam as 98.5 to 99.5%, suggesting that all the incoming sand and gravel are trapped and only some fine-grained sediments pass through. No significant bed lowering, however, was evident in the assessment of Erksine et al. (1993): both localised contraction through bar and bench formation and erosion on the outside of bends was found.

A review of initial VEFMAP results between 2009 and 2013 also provided no conclusive evidence, with erosion and aggradation both present (data supplied by Geoff Earl, GBCMA). The question of sediment supply is one of the efficacy of tributaries downstream of Eildon Dam in supplying coarse-grained sediments, relative to elevated sediment transport potential of the main channel.

Considering the limited number of sites with coarse-grained sediment, and the small variations in sediment smothering, the lack of supply of coarse-grained sediment may be a major limiting factor in the condition of substrate sediment. The historic changes in bedload sediments in the Goulburn River are poorly understood, including the extent to which the Goulburn River should comprise an extensive sand/gravel bed and bars.

Further research

The role of environmental flows in managing substrate sediment condition is uncertain due to inherent variability in sedimentation at a site and the time consuming nature of data collection and analysis leading to limited data. The understanding of physical habitats, such as slackwaters and deepwater habitat, is more certain but verifying these explicit relationships to discharge would further assist water managers. A number of opportunities exist to improve understanding of physical habitat and environmental flows including:

- Quantification of hydraulic conditions (velocity, shear stress) at sampling points for the full range of managed discharges (rather than simple inundation depth metrics that may be appropriate for assessment of ecological condition such as vegetation). Understanding geomorphic processes is dependent on hydraulic data over a large range of discharges, and these data are most achievable through two-dimensional hydraulic modeling of a site. In particular antecedent hydraulic conditions are most likely responsible for conditions at a point in time, rather than measurements made at a couple of points in time. Hydraulic model data will not only benefit physical habitat and sediment smothering analyses, but may also be useful for ecosystem processes such as nutrient dynamics.
- Field testing of hydraulic models using an ADV (rather than an ADCP) for reliable understanding of shallow-water hydraulics.
- Statistical analysis of physical habitat variation relative to environmental flows, based on hydraulic information (as recommended above). For example sediment smothering could be quantified with respect to cumulative antecedent shear stress, and related to flow management.
- Better defining repeatable cross-sections (VEFMAP), clearly marked on both banks for accurate repeat surveys.
- Ascertaining acceptable levels of sediment smothering in the bed for a range of biota, on both a temporal and spatial scale.
- Quantifying links between suspended sediment and environmental flows (as opposed to catchment-derived flows) based on automated suspended sediment sampling.
- Quantifying links between environmental flows, bank erosion (notching, slumping) and suspended sediment concentrations that influence sediment smothering and water quality.
- Quantifying the impact of the Eildon Dam on the trapping of coarse-grained sediments, and the role this might play in the availability of substrate habitat.

7 NUTRIENTS AND DISSOLVED OXYGEN

7.1 Evaluation question and expected outcomes

This section addresses the evaluation question: How are nutrient status and dissolved oxygen levels affected by flow manipulations? The expected outcome was that environmental flows would prevent occurrence of sustained low dissolved oxygen conditions in the Broken Creek weir pools.

7.2 Component Activities - background

Low oxygen in flowing systems is often attributed to microbial activity in response to the addition of black water to river and stream channels (Whitworth *et al* 2012). However, previous work in Rices Weir on Broken Creek indicated that sediment oxygen demand is the dominant oxygen-consuming process (Rees *et al*. 2007). Modified flows, though environmental water, are a mechanism that can be used to increase the oxygen levels in Rices weir.

Detailed study of the sediment processes in Rices weir are limited to a study over the summer of 2007, but this work, combined with other activities of the managing agencies have led to improved monitoring and management of water quality in Rices weir. Broken Creek also underwent a major flooding event in early 2012, potentially altering the nature of the sediment in the lower weir pools. Little previous work has examined upstream weir pools.

This component of the Broken Creek study examined how various factors would lead to oxygen changes in the Broken Creek and whether Commonwealth environmental water led to improved water quality in Rices weir. To this end, we examined how oxygen and nutrients responded to flow over the period from June 2012 through until April/May 2013

7.3 Sites and Methods

A majority of the work in this component was carried out in Rices weir, the most down-stream site on the lower Broken Creek. Some comparisons were made between Rices and Kennedys weir, the weir pool immediately up-stream of Rices weir. In addition, we also compared how sediment carbon and nutrients varied between Rices, Kennedys, Hardings and Balls weirs.

Water column dissolved oxygen. Water column dissolved oxygen was obtained from four sources:

1. Long-term dissolved oxygen (DO) concentrations were obtained from *in situ* probes in Rices weir, with data downloaded from the Victorian data warehouse.
2. Four continuous logging DO probes (D-Opto probes, Zebra-Tech, NZ; accuracy 1% or 0.02 ppm whichever is greater; resolution 0.001 ppm), were deployed over a vertical profile in Rices Weir, positioned to measure DO at 35 cm, 65 cm, 93 and 118 cm depth. In general, the probe at 118 cm depth was 10-20 cm above the bottom of the weir pool. The DO probes were positioned approximately 20 m upstream of the weir wall. The DO vertical-profiling chain was deployed during Nov 2012 to coincide with deployment of continuous water velocity-monitoring devices (described in section 8).
3. Two continuous logging DO probes were deployed on a profile chain in Kennedys weir. The top probe was positioned approximately 20 cm under the surface while the bottom probe was positioned at approximately 1.5 m below the surface, logging DO at the bottom of the weir pool.
4. Spot profile measurements were made in Rices and Kennedys weirs during six field trips (3-6-12, 14-8-12, 18-9-12, 23/24-10-12, 14/15-11-12, 30-1-13).

Dissolved oxygen profiles (top and bottom of weir pool) were also obtained from DO loggers deployed adjacent to the Rices weir wall by Goulburn Murray Water and Goulburn Broken CMA.

Sediment oxygen demand (SOD). Between 8 and 10 closed chambers, (20 cm diameter, operating volume 2.4 L) fitted with logging DO probes and circulating water pumps (Rees *et al*., 2005) were deployed by pushing the chambers 3 cm into the bottom sediment at random locations throughout Rices Weir. DO within the chambers was measure continuously for 2-4 hours. Control chambers measured oxygen consumption in the water column.

Temperature profile. A thermistor chain was constructed using HOBO pendant temperature/light loggers (Onset computer corporation, USA), attached to a float so depth below surface remained constant and deployed adjacent to the DO probes. The pendant loggers were positioned at approximately 10 cm intervals and logged temperature every 10 minutes. Water depth was approximately 1 m, but varied, depending on flow. Temperature data were also collected by the DO probes.

Sediment carbon content. Sediment carbon content was measured in the top 1 cm of 10 mini cores (3 cm diameter) taken randomly throughout the weir pools. Organic content of samples was determined by measuring the mass loss on ignition (LOI) at 550 ºC for 2 hours.

A series of cores, up to 80 cm in depth were taken from Rices and Kennedys weir pools during the May 2013 field trip. Five cores were randomly taken throughout Rices weir and included three cores approximately 5-8 m from the edge and two from mind-channel. The cores in Kennedys weir were taken from the mid channel region.

Chemical analysis. Long-term nutrient data for the period 2012 – 2013 in Rices weir were obtained from the Victorian data warehouse. Total nitrogen obtained from the data warehouse was measured as Total Keldahl Nitrogen (TKN).

Spot measurements of nutrient concentration were made by taking samples at four weir pools along the length of the lower Broken creek. Five replicate samples were collected throughout the weir pool. Dissolved nutrient samples were filtered through 0.45 µm-pore sized membrane filters. All samples were frozen prior to analysis. Standard methods of nutrient analysis were used throughout. Total nitrogen (TN), total phosphorus (TP), oxides of nitrogen (NOx), ammonium and filterable reactive phosphate (FRP) were measured according to national standards of quality control and quality assurance at the Murray-Darling Freshwater Research Centre analytical chemistry laboratory, which operates under national accreditation (NATA accreditation).

Spot measurements of DO, turbidity, pH and salinity were carried out using a hand held multi-probe meter system (Aqua lab). Profiles were taken by submersing the probe through 10cm intervals and allowing measurements to stabilise prior to recording the results. Ad hoc spot measurements were made of surface water quality throughout the weir pools.

Water column chlorophyll-a (Chl-a). Water samples were filtered through glass-fiber filters (Whatman, GFC filters) and the chl-a measured by extracting the filters in hot ethanol for 10 minutes (ISO 10260, 1992).

7.4 Results

Water column dissolved oxygen – Rices weir

Typical plots show the diurnal oxygen concentration often varied by up to 2 mg L^{-1} during summer (Figure 7.1). Only minor variation in DO levels through the water column occurred during the period shown, which can be seen as the probe at depth always measuring slightly lower dissolved oxygen than the probes at the upper levels. There was a clear difference in the DO at the top and bottom of the Creek towards the end of the period shown.

High daily variation obscured any long-term relationship between discharge and DO (data not presented). Improved daily DO, associated with increased discharge throughout the study period (Figure 7.2) are consistent with an argument that Commonwealth water was successfully used to maintain DO levels in Rices weir.

Figure 7.2 identifies (with arrows) a series of occasions where DO increases very shortly after, or concurrent with an increase in flow. It is important to note that the improved DO in response to flow is not necessarily linear and the reasons for this deviation are examined in section 8. Furthermore, any improved DO was very short lived. In general, the average daily DO showed a strong decline over summer, with Simple regression analysis of DO vs. flow showed no relationship (data not presented). In further analyses we examined how the rate of change in the 1, 2 and 3-day average of DO responded to the rate of change in the 1, 2 and 3-day average of flow. These durations were chosen as our general understanding of the rapid response of algal and flow dynamics in Rices weir indicated we might see clear responses in this short frame. Similar results were obtained with each analysis so we present only the 3 day average (Figure 7.6). The large spread of points showing change in DO even when there was no change in flow (x-axis) demonstrates that DO is responding to other factors, for example, instantaneous algal dynamics or temperature.

Historical records show that DO in Rices weir can fall to levels that are known to be harmful for fish (Figure 7.4). A further example of low DO occurred during early 2012 (Figure 7.5). During this period a low DO event occurred in February, followed by a flood during March, which was accompanied by an extended period when high dissolved organic carbon (DOC) was present in the creek.

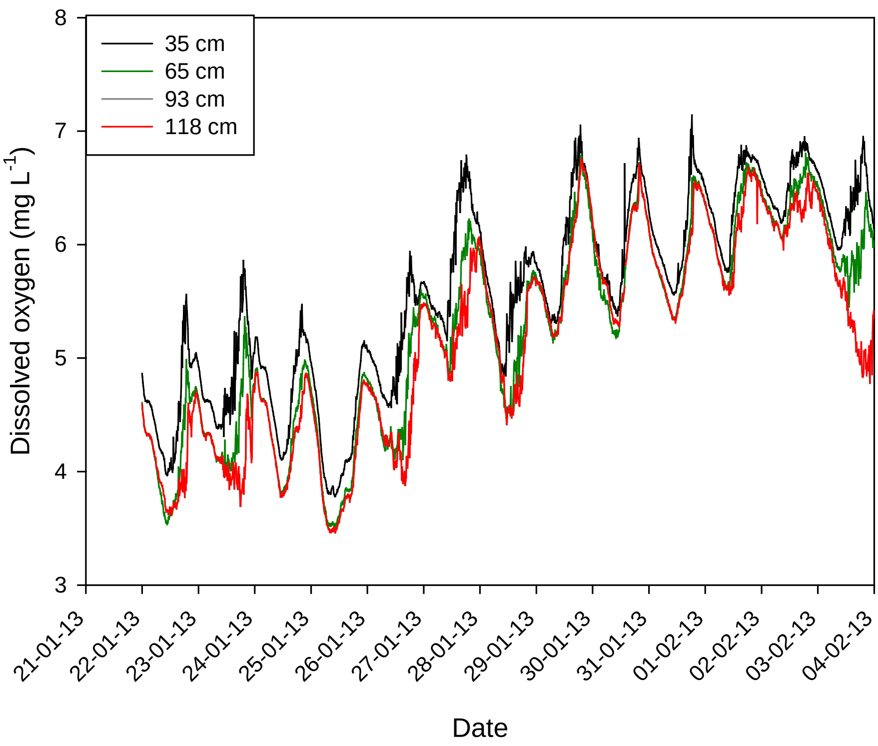


Figure 7.1: Typical dissolved oxygen recorded at four depths in Rices weir, in this case from 22 Jan 2012 to 4 Feb 2013. The depth is measured from the water surface.

Simple regression analysis of DO vs. flow showed no relationship (data not presented). In further analyses we examined how the rate of change in the 1, 2 and 3-day average of DO responded to the rate of change in the 1, 2 and 3-day average of flow. These durations were chosen as our general understanding of the rapid response of algal and flow dynamics in Rices weir indicated we might see clear responses in this short frame. Similar results were obtained with each analysis so we present only the 3 day average (Figure 7.6). The large spread of points showing change in DO even when there was no change in flow (x-axis) demonstrates that DO is responding to other factors, for example, instantaneous algal dynamics or temperature.

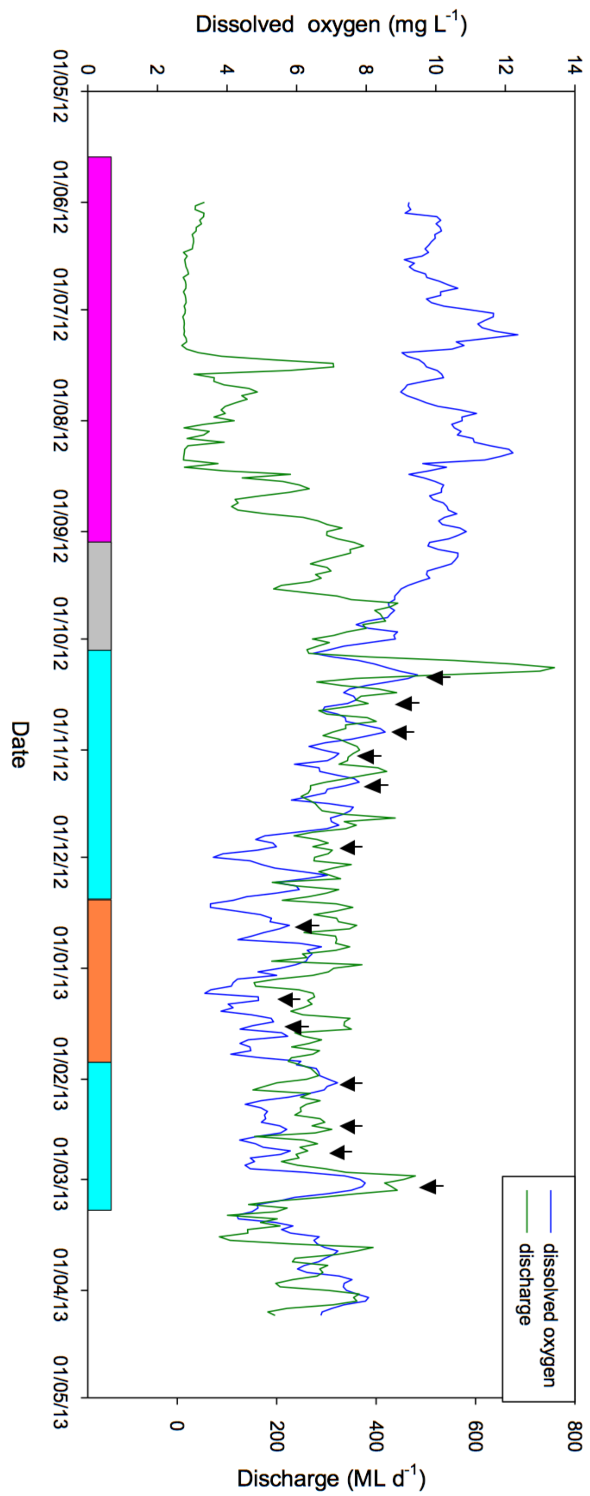


Figure 7.2: Average daily water column dissolved oxygen and discharge in Rices weir June 2012 to April 2013. The period represented by the pink bar shows unregulated flow in Broken Creek. The grey section shows combined unregulated flow and Commonwealth environmental water. Cyan shows the period with only Commonwealth environmental water and orange shows when Commonwealth environmental water was supplemented by inter-valley transfers. Arrows highlight specific increases in DO that follow in response to some increase in discharge.

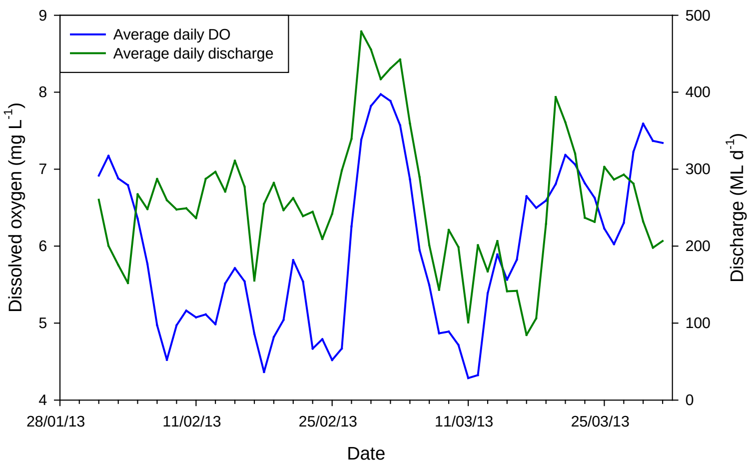


Figure 7.3: Water column dissolved oxygen and flows from 28 Jan 2013 to 28 March 2013.

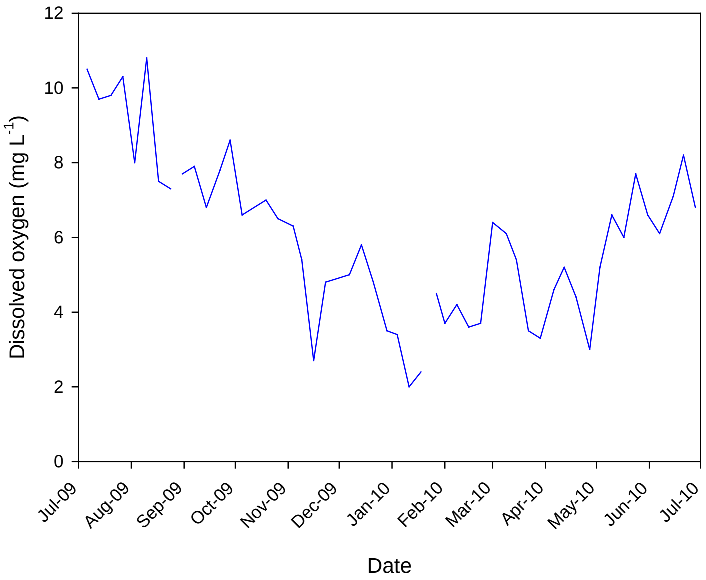


Figure 7.4: Dissolved oxygen in Rices weir, July 2009 - July 2010.

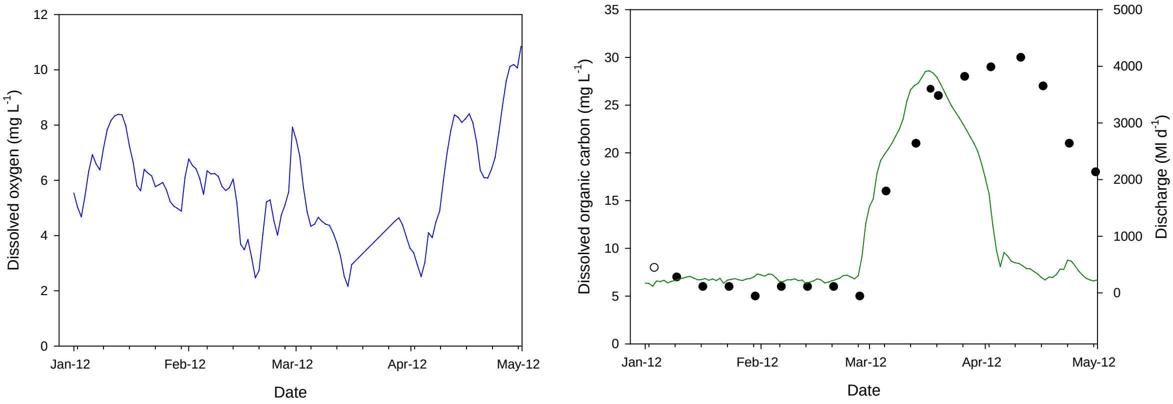


Figure 7.5: Dissolved oxygen (left panel) in Rices weir, January 2012 to May 2012. Right panel shows dissolved organic carbon and discharge (green line).

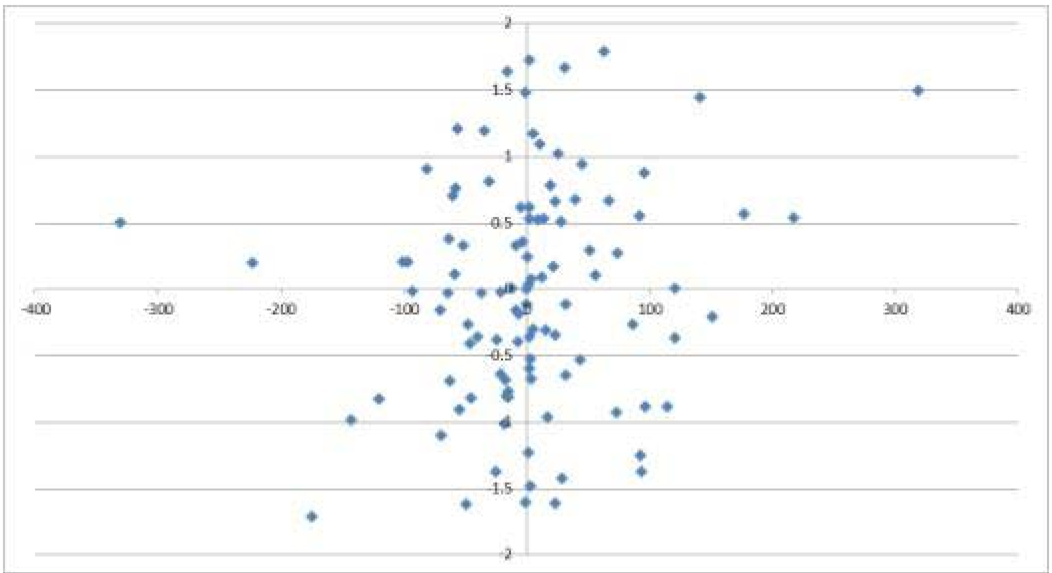


Figure 7.6: Average change in dissolved oxygen over 3-days (y-axis) plotted against average change in flow (x-axis) over 3-day.

Water column dissolved oxygen – Kennedys weir

In general terms, the trends in DO in Kennedys weir did not show any great difference from those in Rices weir. For example, DO concentration and trends over time, and the range in the diurnal variation that occurred during January 2013 were very similar at both the sites (Figure 7.7). Shortly after this period however, there was variation in DO between upper and lower waters. Daily oxygen production is likely to be considerably higher in the upper levels of the creek (seen by the sharp peaks of the black lines - Figure 7.8), while lower algal production at depth led to a lower DO and smaller daily variation (red line, Figure 7.8).

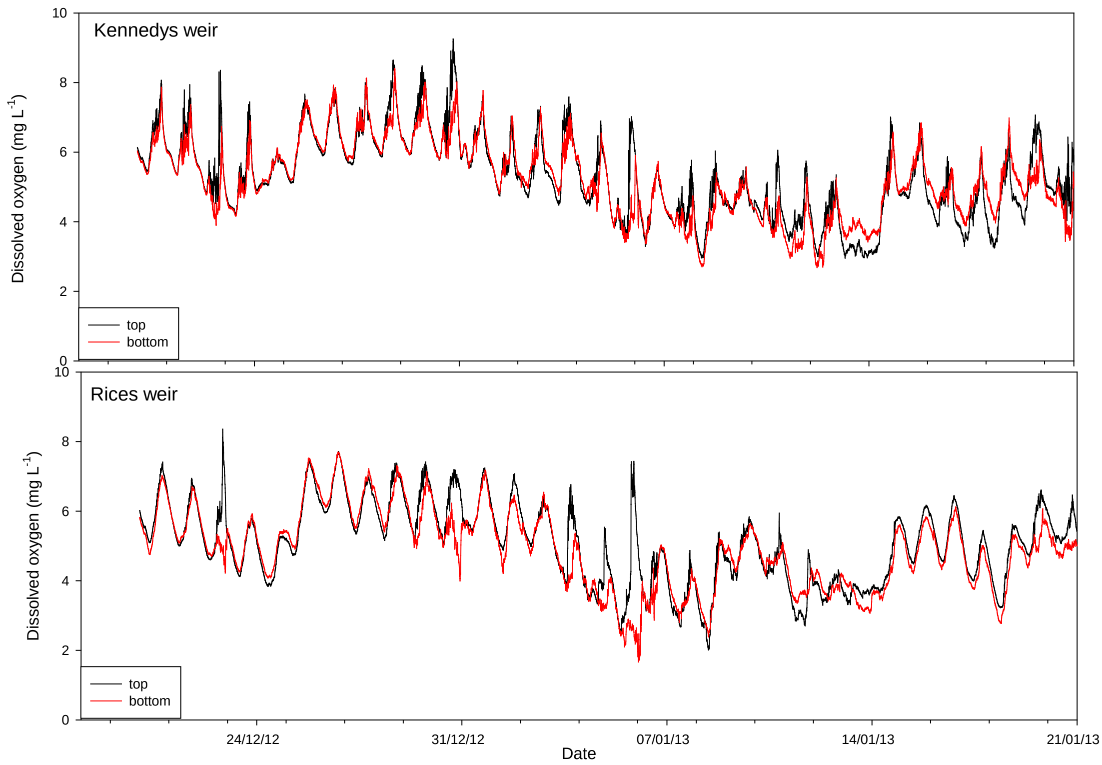


Figure 7.7: Dissolved oxygen in the upper and lowest level of Kennedys weir (top panel) and Rices weir (bottom panel)

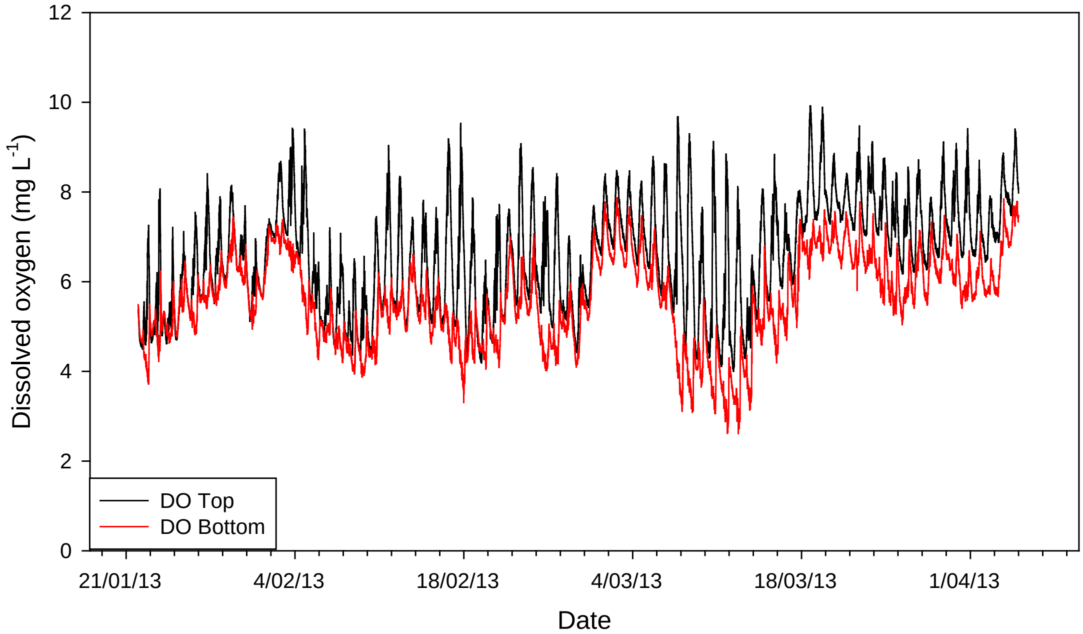


Figure 7.8: Daily dissolved oxygen concentration in the upper level and bottom of Kennedys weir pool.

Sediment oxygen demand (SOD):

Sediment oxygen demand varied considerably over the study period (Figure 7.9). Low values were measured during the winter periods, but the anticipated increase during summer, due to higher temperature, was not detected in this study. Instead, high variability obscured any patterns that might have existed. There also was no consistent difference between the sediment oxygen demand in Kennedys weir and Rices weir.

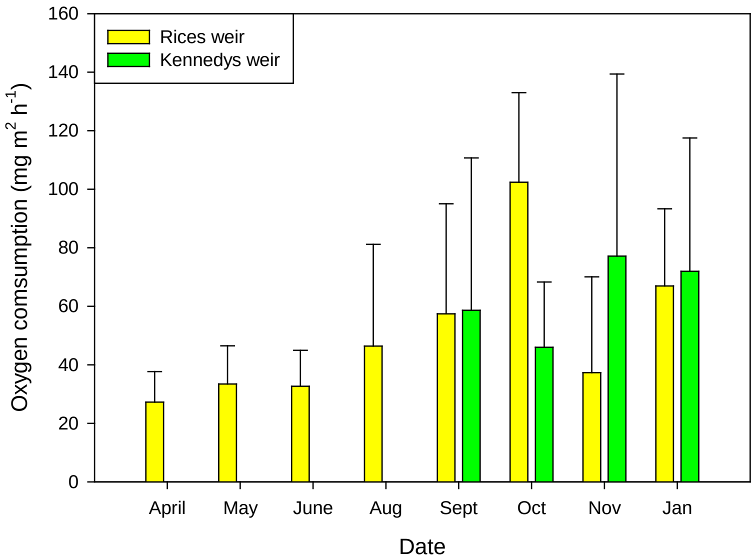


Figure 7.9: Sediment oxygen demand in Rices and Kennedys weirs. Error bars indicate standard deviation.

Sediment organic matter content

The amount of organic material in the top 3 cm of sediment showed no significant change over the study period covered in this report (data not presented). All data subsequently were pooled and the average amount of organic material in the sediments in each of the weir pools were compared (Figure 7.10). The amount of organic matter in the two up-stream sites was not different from each (approximately 4% at both sites), but were significantly lower than the organic content at both the downstream sites (Kennedys and Rices), where organic content was between 8 and 10% of the sediments.

The variability in the amount of organic matter in the sediments of Rices and Kennedys is further highlighted by the sediment core organic content profiles (Figure 7.11, Figure 7.12). Individual cores each showed a slightly different profile, until the base clay level was reached where carbon content was general around 4.5% across all cores. The peak organic content was generally around 10-11%, but this did not always occur at the top of the profile. For example, core 1 in Rices weir showed a deposition layer approximately 5-10 cm below the surface, while core 3 shows a deposition layer at 20 cm below the surface. Core 5 was taken mid-stream and it shows a deposition layer (9.2% organic content) at approximately 70 cm below the surface.

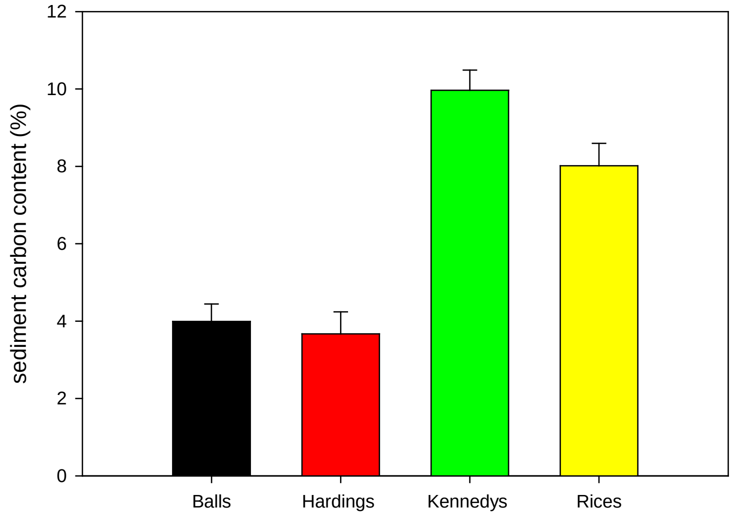


Figure 7.10: Organic content in the sediments of four weir pools in the lower Broken Creek. Samples were taken throughout the study period and pooled for the analysis. Error bars show standard error.

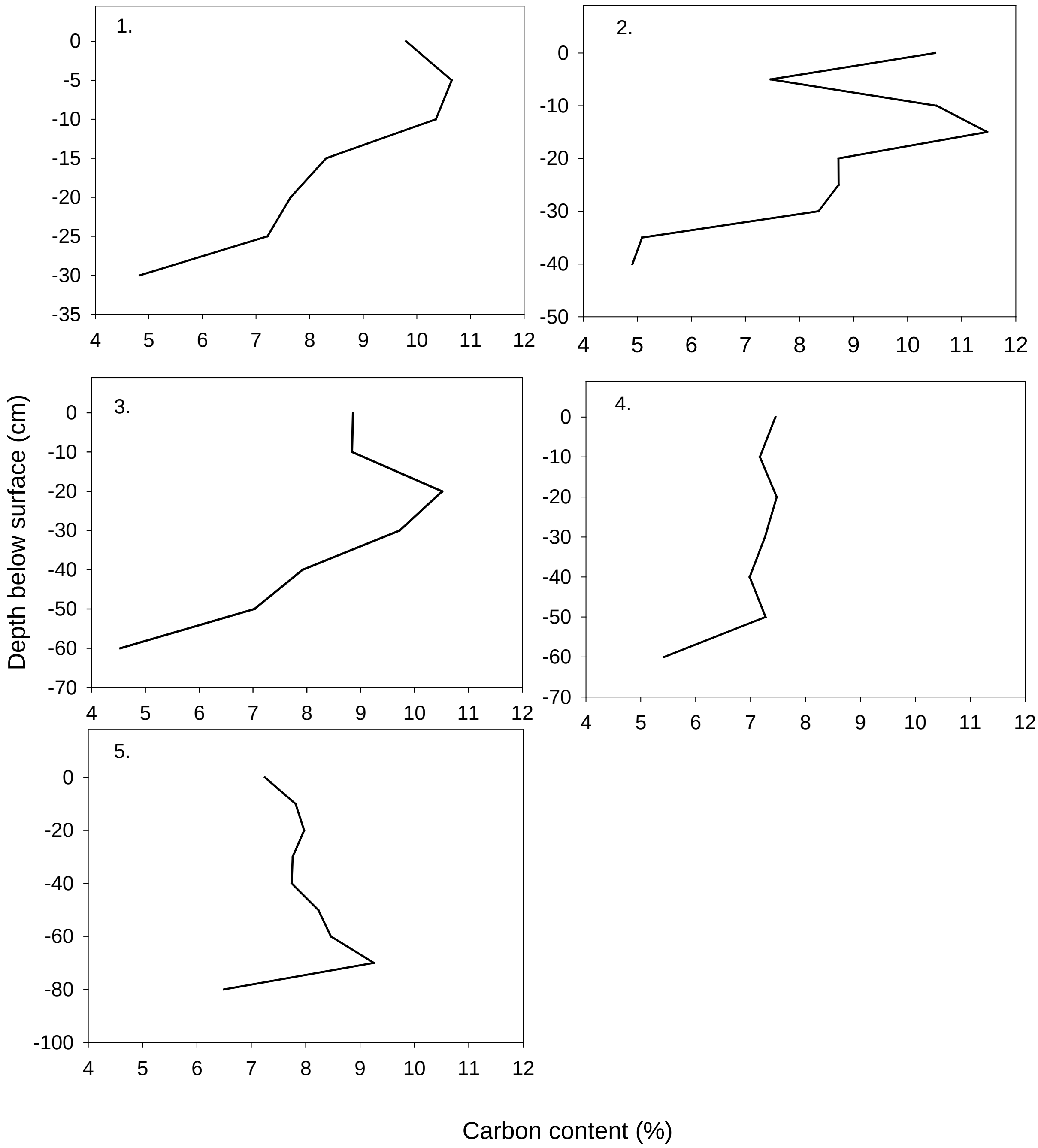


Figure 7.11: Carbon content at depth in the five sediment cores from Rices weir. The difference in scales of the y-axes reflects the variability in the depth before the base clay levels were cored. No attempt was made to core deeper into the base clay layer.

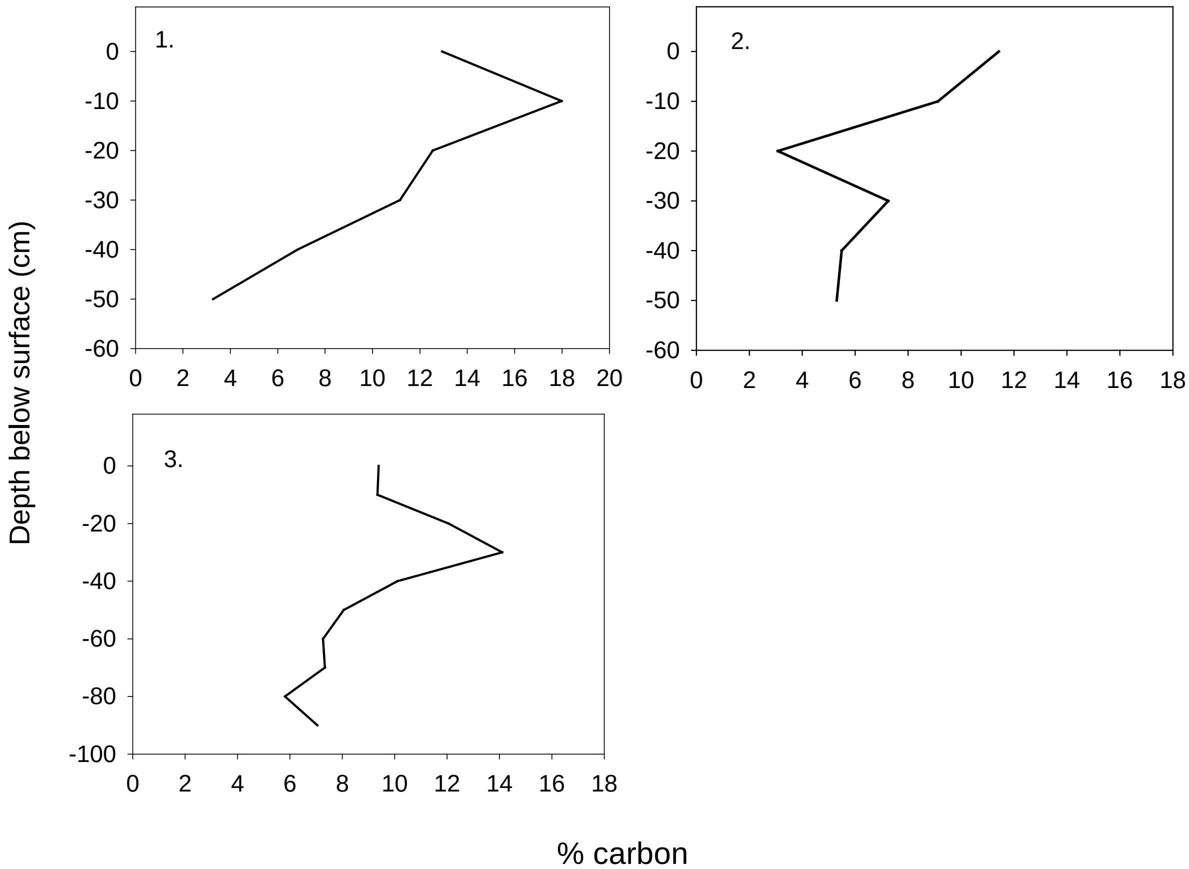


Figure 7.12: Carbon content at depth in three cores from the sediments of Kennedys weir. The difference in scales of the y-axes reflects the variability in the depth before the base clay levels were cored.

Response of carbon, nutrients and water quality in Rices weir

<u>Phosphorus species</u>. Total phosphorus (TP) broadly showed a trend throughout the year, increasing from an initial level of approximately 250 mg L^{-1}, rising to approximately 500 mg L^{-1} in summer before declining again (Figure 7.13). A notable spike in TP occurred during August 2012. Patterns in the concentrations of filterable reactive phosphorus were less obvious, although appear to have increased to maximum during January and February 2013.

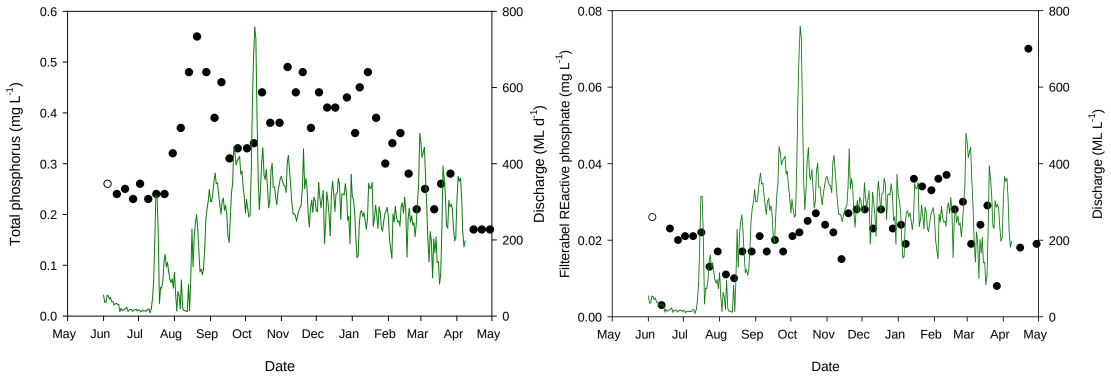


Figure 7.13: Total phosphorus (left panel) and filterable reactive phosphorus (right panel) in Rices weir May 2012 to May 2013. The green line shows average daily discharge thought the year.

Nitrogen species. There was considerable weekly variation occurred in the Total Keldahl nitrogen (TKN) of Rices weir, making it difficult to see any pattern throughout the year (Figure 7.14). A major increase in TKN occurred during August 2012, mirroring exactly the increase seen in TP Figure 7.14. NO$_x$ varied considerably throughout the year and seemed to show little in the way of a seasonal response. Occasional ‘spike values’ in NO$_x$ were measured, particularly during August 2012. However, the increase was not consistent and reasons for the increase remain unknown at this stage.

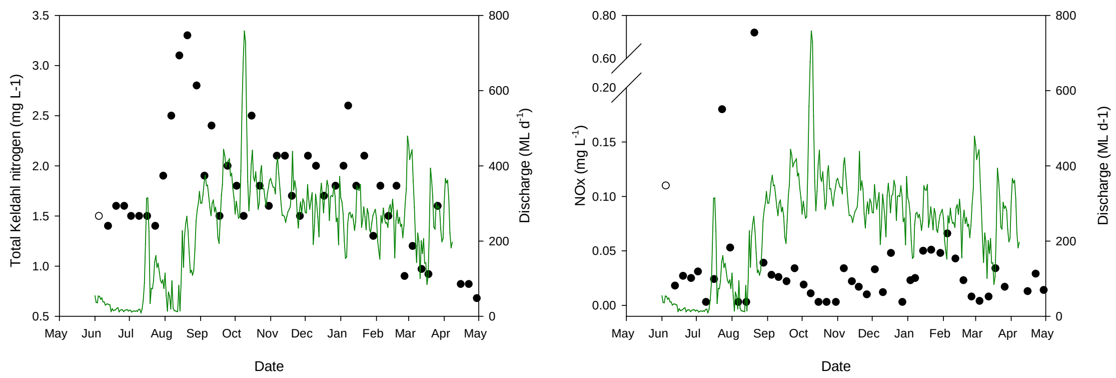


Figure 7.14: Total Keldahl nitrogen (TKN) (left panel) and Oxides of nitrogen (NO$_x$) (right panel) in Rices weir May 2012 to May 2013. The green line shows average daily discharge thought the year

Dissolved organic carbon. DOC during May 2012 was between 9 and 10 mg L^{-1}, consistent with Broken Creek still receiving drainage from the floodwaters that were surrounding the Creek, from the flood that had occurred in March 2012 (Figure 7.15). A large increase occurred during August, followed by a decline throughout September and October, after which DOC remained between 6 and 8 mg L^{-1} throughout the remainder of the study period.

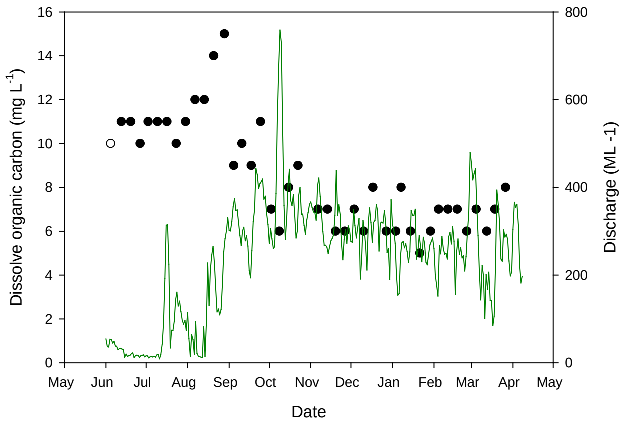


Figure 7.15: Dissolved organic carbon in Rices weir May 2012 to May 2013. The green line shows average daily discharge thought the year.

Turbidity. Turbidity increased throughout the summer period, rising to a maximum in January (270NTU), before declining again to approximately 90 NTU (Figure 7.16).

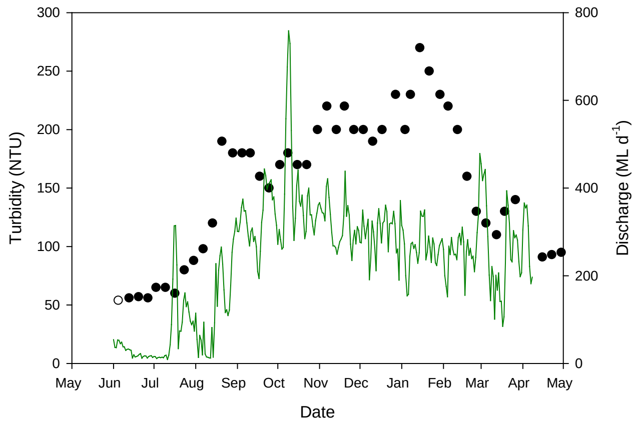


Figure 7.16: Turbidity in Rices weir May 2012 to May 2013. The green line shows average daily discharge thought the year.

Longitudinal response of nutrients

The overall observation from the six field trips to the Broken Creek is that site-specific factors played a dominant role in determining the concentrations of nutrients in the four weir pools, and that the length of the lower Broken creek is insufficient to drive simple longitudinal increases in nutrient concentrations (Figure 7.17, Figure 7.18). Notable observations include peaks in filterable reactive phosphorus (FRP), NOx, ammonium and TN during August 2012;

the soluble N species showing dramatic increases. There were periods when Balls weir had significantly higher FRP and NOx than the downstream sites. This may be reflecting inputs from Nathalia or Murray Valley drain #13.

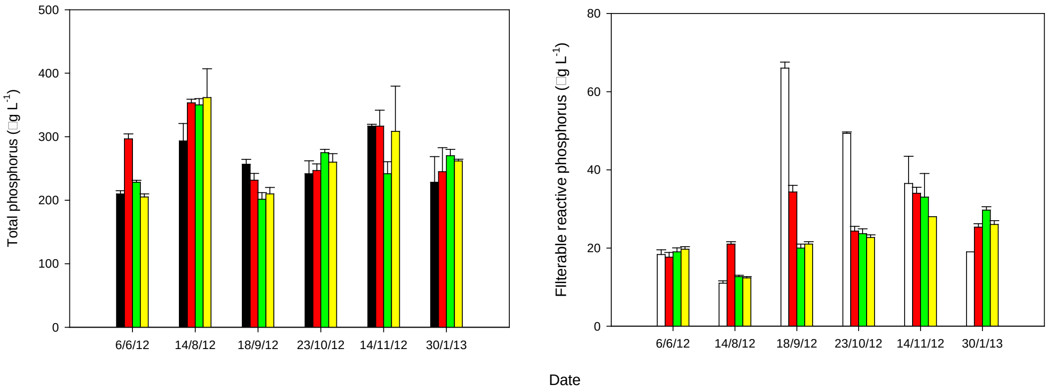


Figure 7.17: Total phosphorus (left panel) and filterable reactive phosphorus (right panel) in four weir pools of the lower Broken Creek. Black bars (Balls weir), red bars (Hardings weir), green bars (Kennedys weir), yellow bars (Rices weir)

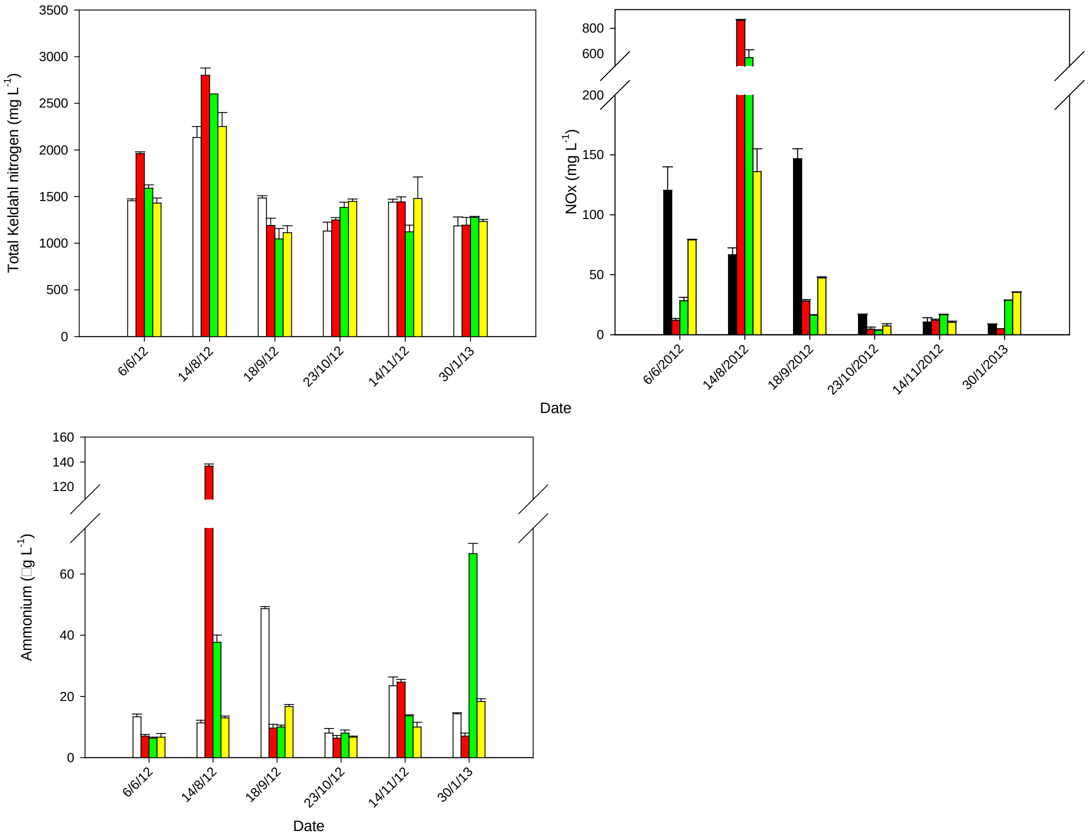


Figure 7.18: Total Keldahl nitrogen (upper left panel) oxides of nitrogen (NO_x) (upper right panel) and ammonium (lower left panel) in four weir pools of the lower Broken Creek. Black bars (Balls weir), red bars (Hardings weir), green bars (Kennedys weir), yellow bars (Rices weir).

Chlorophyll –a content

Chlorophyll–*a* was very high in both weir pools, ranging from 50 to 130 µg L^{-1} (Figure 7.19). Unfortunately samples were lost during processing and so the incomplete data set does not allow for any real comparisons over time and between sites. Be that as it may, the very high chlorophyll levels demonstrate very high algal biomass present in the creek, leading to high photosynthetic activity in the water column, explaining the high daily variation in the DO concentrations.

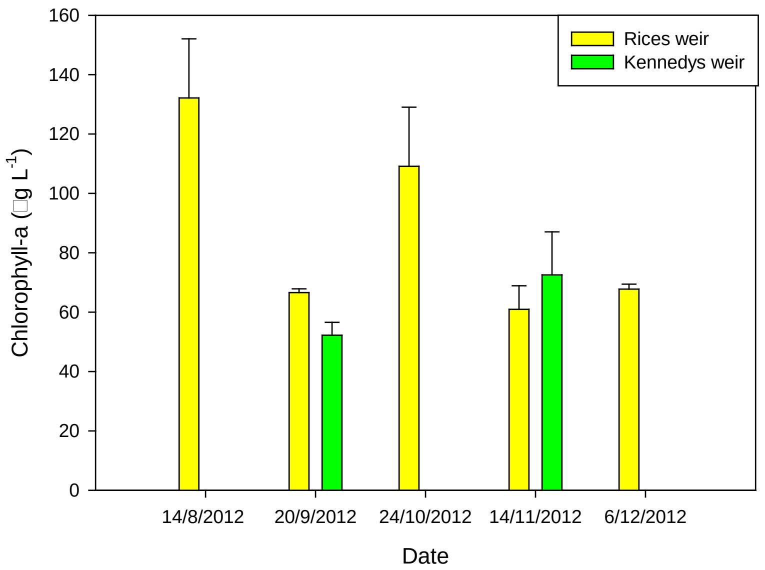


Figure 7.19: Chlorophyll-a in Rices and Kennedys weir.

7.5 Discussion

Previous accounts report that dissolved oxygen in Rices weir pool can decline to extremely low levels (Rees *et al*. 2007) and fish deaths in 2002 were attributed to low DO concentration. DO has not been recorded with sufficient frequency over the longer term to ascribe cause and effect of low DO, but higher frequency monitoring (Rees *et al.* 2007) showed strong vertical gradients in DO could occur and that oxygen concentrations could be close to zero at depth. Installation of high-frequency DO probes and management by Goulburn Murray Water and the Goulburn Broken CMA have been used to monitor *in situ* DO, and to respond to low DO events by discharging flushes of water down the creek.

Hypoxic water events in riverine systems can often be attributed to flow events mobilizing DOC, which is decomposed by bacteria, which also consume oxygen during the decomposition process (Hladyz *et al*. 2011, Whitworth *et al*. 2012). Broken Creek flooded in early 2012, leading to an extended low DO event coinciding with large amounts of DOC being deposited in the creek. The low-lying nature of the landscape meant drainage to the creek occurred for some time. Data in this study show some enrichment of DOC occurred right through into May and June 2012 showing some effect of the earlier flood was still apparent. The DOC had returned to 'typical' levels well before summer, when dissolved oxygen started to decline in Rices weir, and this decline is due to the conditions within the weir pool, rather than any external enrichment of DOC.

During our study, DO within Rices weir often varied up to 2 mg L^{-1} over 24 hour periods. These variations are similar to those reported previously in Rices weir (Rees *et al.* 2007). These short-term variations in DO obscured any simple relationships between DO and discharge and simple regressions could not be used to examine relationships between DO and discharge. However, by examining individual events throughout the summer, there is evidence consistent with an average daily DO in response to increased discharge. It is important to note that there were some occasions where the relationship between DO and discharge was not as clear and the extent of any change in DO did not necessarily respond in a linear fashion to a change in discharge (Figure 3.2). This was highlighted by some preliminary comparisons of the rates of change in DO and flow, which showed there were periods when DO changed although there were no changes in the rates of change of flow. Further analysis examining whether there were delayed responses in rates of change of DO also showed no clear relationship (data not presented). These data indicate that a more complex mathematical modeling approach is required to account for changes that may be occurring. Such an approach is presented in Section 8.

Nutrients showed little response to flow. Weekly nutrient sampling occurred in Rices weir, which did enable us to look for very general trends over the summer, but the sampling time frequency was not sufficient to address specific responses to flow. A large pulse of DOC and nutrients occurred in August, but this was well past the flood period. No other data are available on the source of the DOC as comprehensive up-stream sampling was not carried out, and the origin of the DOC remains speculative.

Sediment oxygen consumption was highly variable during this study. While consecutive low values were obtained during the winter periods, there was large variability within sites, evidenced by the large error bars in the sediment oxygen data for each of the field trips. Sediment oxygen demand in Kennedys weir was generally very similar to Rices weir. The sediment carbon levels typically ranged from 9 to 11% within the replicate samples, which would not necessarily be considered a great difference. We can only suggest the nature of the carbon was different, with considerable variability in the bioavailability of the sediment carbon throughout both weir pools.

Previous reports of sediment oxygen consumption in Rices weir ranged from 66 to 300 mg O_2 m^2 h^{-1} over the summer period (Rees *et al*. 2007). During our current study some individual chambers had oxygen consumption rates up to 180 mg O_2 m^2 h^{-1} (data not presented). However, the average rates of oxygen consumption across all chambers were at the lower end of those of the previous work, and we did not measure oxygen consumption equivalent to the most extreme values of the previous work. Despite the apparently lower rates, these values still represent oxygen consumption rates between 2 and 5 times those that have been reported for the Murray and Ovens rivers (Rees *et al*. 2007). Strong vertical oxygen gradients did not occur during the summer, but there were times when the DO was decreased at depth. We suggest three possible reasons for this: 1), there was a real decrease in the sediment oxygen demand during the summer of 2012-2013, leading to reduced draw-down of DO at depth, 2) natural annual variability accounts for the difference seen in previous years, 3) current real-time DO measurement and flow management regimes are more refined than those used in the past, meaning water managers can respond more effectively to any low DO events.

Sediment carbon content in the top few cm of Rices weir is not appreciably different from previous years. This study is the first to measure carbon profiles in the sediments of Broken Creek, and the cores showed that a high a carbon content was present for quite some centimeters below the surface. Interestingly, the cores also showed major deposition layers at depth. Any deposition layer will be the balance between deposition and mineralisation. Mineralisation will be very slow (if measurable at all) at depth and given one core contained a peak in carbon content 80 cm below the surface, some historical deposition of carbon is very likely to have occurred. The carbon layers are consistent with accounts of Azolla blooms, but the exact nature of the carbon has not been determined in this study. Examples of Azolla blooms in the last 10 years are well documented, but earlier accounts are always sketchy and frequently obscured by human memory. If future study can show the nature of the carbon layer and its age then we may be able to cast more light on the historical importance of Azolla blooms.

The absence of long-term data prevent us from describing true effect of Commonwealth environmental water on DO in Rices weir based on the methods in this component of the program. However, given that 1) there were similar carbon loads to previous years, 2) there were similar sediment oxygen demands in Rices weir (albeit slightly reduced from previous years), 3) DO changes mirrored specific flow manipulations, and 4) historical data indicate low flow events in Broken Creek lead strong draw-down of DO, it is reasonable to suggest that the Commonwealth environmental water had a major role in maintaining DO at levels suitable for biota to survive. This question is examined further in the next Section.

8 THE EFFECT OF ENVIRONMENTAL FLOW ON DISSOLVED OXYGEN

8.1 Evaluation question and expected outcomes

This section addresses the evaluation question: Have environmental flows contributed to the maintenance of high dissolved oxygen levels in Broken Creek weir pools? The expected outcome was that environmental flows promote re-aeration of the water column and avoid low oxygen conditions.

8.2 Component Activities - Background

Water quality concerns in the Broken Creek at Rices Weir have centered on low dissolved oxygen concentrations (DO) with associated effects including increased risk of fish mortality, altered biogeochemical cycling and eutrophication as nutrients are released from sediments. Eutrophication is a particular problem because it can increase the biomass and the coverage extent of *Azolla*, a floating fern. As the *Azolla* dies and settles, it is thought to form organic-rich sediment with a high sediment oxygen demand that stimulates recycling of carbon and nutrients into the water column, contributes to maintenance of low DO and thus perpetuates the conditions for *Azolla* to thrive (Rees *et al*. 2007). Thermal stratification during periods of low flows inhibits mixing of the water column and promotes hypoxic conditions in bottom waters (Bormans & Webster, 1997; Whiterod & Sherman, 2012).

Environmental flows in the lower Broken Creek are specifically targeted at:

1. maintaining DO above 5 mg/L (within ANZECC guidelines), and

2. minimising the growth of *Azolla* (CEWH, 2011).

Increased flow could flush *Azolla* from the system, thus reducing sediment deposition. Increased flows can also promote reaeration at the water surface and increase turbulent mixing, reduce stratification and decrease the duration of any hypoxic events, sediment nutrient release, and thereby prevent the growth of *Azolla*. Our aim is to set up a simple dissolved oxygen model to assess the impact of environmental flows on the DO concentrations in the lower Broken Creek during the summer of 2012-13.

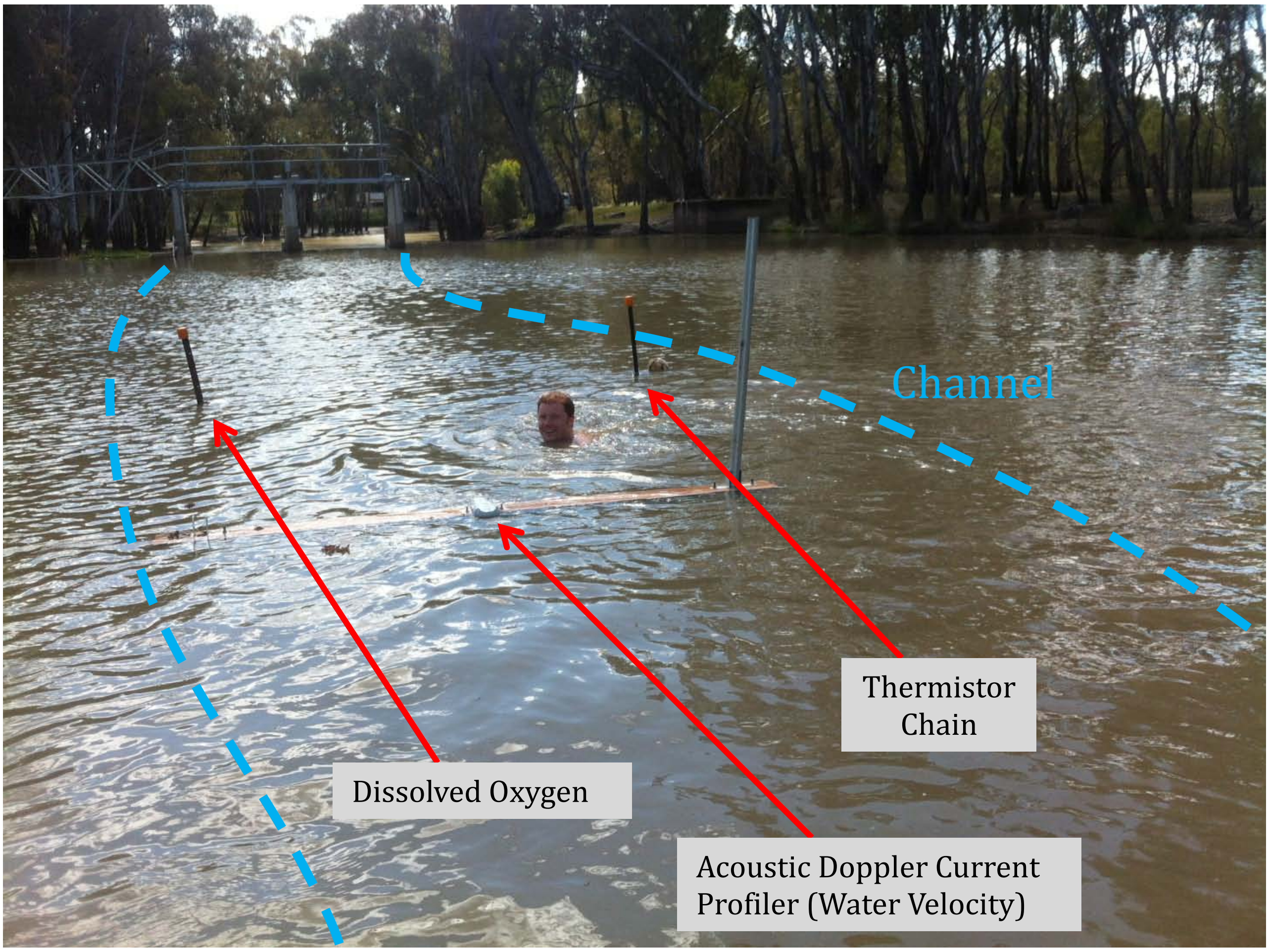


Figure 8.1: Positioning of instrumentation in Broken Creek upstream of Rices Weir (source: R. Young, 25 October 2012)

8.3 Methods

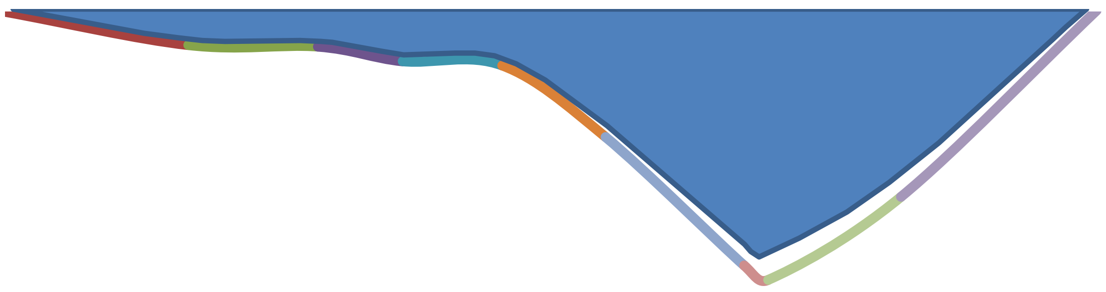
Data collection. Flow velocity profiles were measured in the thalweg of Broken Creek, 15 m upstream of Rices Weir (Figure 8.1), using an IQ-plus Acoustic Doppler Current Profiler (ADCP, SonTek, USA). This instrument uses high-frequency acoustic pulses to measure the Doppler shift from suspended particles flowing through the water column to estimate 2-dimensional water velocity in 2 cm 'cells' from 4 cm below the water surface to 8 cm above the sediment surface. The ADCP was installed on 25 October 2012 and set to record profiles every 15 minutes. From 17 January 2013, the profiling frequency was increased to every 2 minutes in preparation for the proposed flow manipulation experiment. Higher frequency measurements were taken for the remainder of the data collection period, until the 12 May 2013.

Temperature profiles were also measured in the thalweg, approximately 2 metres downstream of the ADCP. HOBO temperature loggers were suspended at 6 depths below the water surface (0 cm, 14 cm, 34 cm, 49 cm, 66 cm, and 87 cm). The loggers were set to measure temperature profiles every 5 minutes. Four HOBO DO sensors were installed at 35, 65, 93 and 118 cm below the water surface to record DO concentrations every 5 minutes. This high resolution of DO measurement was aimed at linking water quality with changes to flow, rather than just looking at the diurnal changes.

Figure 8.2: Meteorological conditions were recorded with this automated weather station from the southern bank at Rices Weir (photo taken on 17 January 2013)

Meteorological conditions, including wind speed and direction, air temperature, relative humidity, rainfall and solar radiation were also monitored from 25 October 2012 using a Campbell's Scientific Automatic Weather Station. From 17 January 2013 a net radiometer was installed in Rices Weir Pool to assess the extent of longwave emission from the water surface and thereby allow an estimation of the irradiance available for photosynthesis (Figure 8.2).

Bathymetry. To assess the bathymetry and cross-sectional area of Rices Weir Pool, 179 depth measurements were taken along ten transects between the high-water mark on opposite banks and at random positions between these transects. These data were interpolated using the Kriging method and plotted in ArcGIS (Figure 8.3). The cross-sectional area of Broken Creek along a 48 m wide transect at the deployed ADCP, was calculated using 10 depths to give the cross-sectional area for estimation of discharge according to measured flow velocity (Figure 8.3).



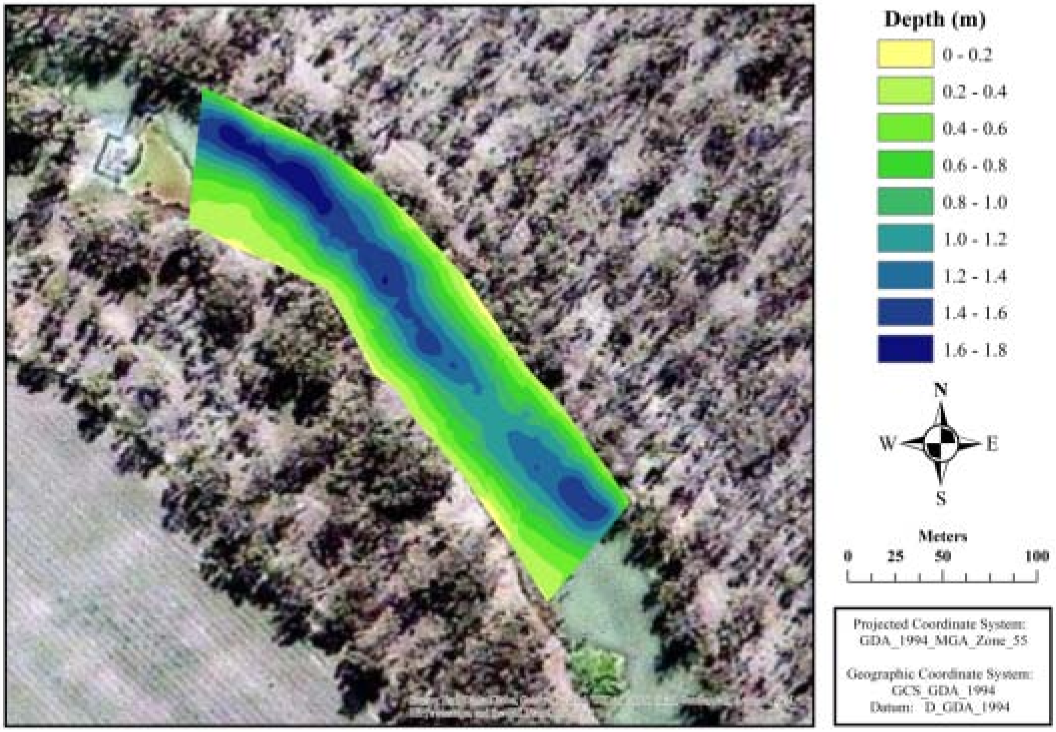


Figure 8.3: Cross-sectional area (top) and bathymetry (bottom) of Broken Creek at Rices Weir

Richardson Number calculations. In conjunction with water velocity profiles, the variation of temperature with depth was used to calculate the Richardson Number (Ri). This provides an indication of the propensity of a water body to mix against any stratified density gradients. Consequently, Ri provides a method of linking any temperature gradient with flow and its calculation provides an initial assessment of the influence of environmental flow delivery. Ri is calculated as the ratio of density forces to the level of advective inertia that destabilises these density differences (equation 1; Linsley *et al*. 1982). As any density differences in Broken Creek are likely to be small because of shallow water depth and consistently low salinity, the Boussinesq approximation is used with a correction for reduced gravity, g' (equation 2).

1. $Ri = \frac{g'L}{U^2}$ (or $Ri = g'.\frac{\frac{dT}{dz}}{\frac{du^2}{dz}+\frac{dv^2}{dz}}$ in discretised form[1])

2. $g' = \frac{\Delta\rho}{\rho}g$

For the calculation of Ri, the water depth is given by *L*, the water velocity is *U*, density is *ρ* and the acceleration due to gravity is shown as *g*. Ri values greater than 0.25 indicate that buoyancy is sufficient to overcome the destabilizing forces from convective flow. Conversely, Ri values below 0.25 suggest that flow is of sufficient magnitude to prevent the formation of temperature stratification.

Oxygen Balance Model. To assess the specific contribution of environmental flow to the maintenance of DO concentrations, a mathematical model of the water column oxygen balance was constructed using modified equations from Uehlinger et al. (2000). The model represents the three primary processes controlling DO levels: (i) oxygen production through photosynthesis; (ii) oxygen consumption through respiration and chemical oxygen demand; and (iii) re-aeration of the water column at the water surface. In this way, the model can be used to simulate oxygen concentrations with and without environmental flow delivery in lower Broken Creek.

The DO balance model is a one-dimensional model representing vertical exchange of oxygen at the water surface and both the generation and loss of dissolved oxygen within the water column and bed sediments. The model assumes oxygen is well-mixed through the water column. The model also assumes there is no net lateral exchange of dissolved oxygen within the weir pool, which is a reasonable assumption given oxygen concentrations downstream through the weir show little variation (Rees, 2013, pers. comm.).

The rate of change in oxygen concentration associated with photosynthesis (P), oxygen consumption (*C*) and re-aeration (*R*) (in units of $kgm^{-3}s^{-1}$) are represented in the model using equations 3, 4 and 5, respectively;

3. $P = \frac{P_{max}Ie^{-k_1T_u}}{(K_P+I)d}$

4. $C = \frac{-E}{d}Q^{(Tb-20)}$

5. $R = \left(K_R + a_1{W_d}^{b_1} + a_2{W_c}^{b_2} + a_3U^{b_3}\right)(C_{sat} - C)e^{k_3(T_s-20)}$

[1] where T_{ref} is the reference temperature, dT/dz is the change in temperature with depth, du/dz and dv/dz are the directional change in water velocity along the x-axis (in line with streamflow) and y-axis (vertically perpendicular to x), respectively, with depth.

In these equations, P_{max} is the light saturated photosynthesis rate ($gO_2m^{-2}s^{-1}$), I is the light intensity at the water surface (Wm^{-2}), K_P is the half-saturation light intensity (Wm^{-2}), T_u is the water turbidity (NTU), k_1 represents the strength of turbidity dependence on photosynthesis, d is water depth (m), E is the ecosystem rate of oxygen consumption at 20 ^oC ($gO_2m^{-2}s^{-1}$), Q represents the strength of temperature dependence in oxygen consumption, T_b is the water temperature at the streambed (^oC), K_R is the re-aeration coefficient in the absence of wind and flow effects (s^{-1}), W_d is speed of wind (ms^{-1}) in downstream direction (i.e. direction of longest fetch and equal to zero if wind is in the upstream direction), W_c is the speed of wind (m/s) of the non-downstream component of wind, U is the depth-averaged flow velocity (ms^{-1}), the parameters a_1, b_1, a_2, b_2, a_3 and b_3 represent the effect of wind and flow velocity on the re-aeration coefficient, C_{sat} is the saturation at actual oxygen concentration (kgm^{-3}) at the mean surface water temperature T_s (^oC), C is the depth-averaged oxygen concentration (kgm^{-3}) and k_3 represents the strength of temperature dependence on re-aeration. We have assumed a constant algal population and the model does not reflect any impact of Azolla, which was not present during the study period. The photosynthesis equation was modified from that of Uehlinger et al. (2000) to account for the effect of turbidity on light availability, the oxygen consumption equation was modified to account for effect of water temperature, and the re-aeration equation was modified to account for wind and flow effects.

The model uses an hourly time-step. Each hour the model calculates the rate of each of the three processes: photosynthesis, re-aeration and oxygen consumption. The sum of these gives the net rate of change of dissolved oxygen through the water column during each hour. Photosynthesis increases with solar radiation producing a diurnal pattern in DO levels. These changes can be summed over each 24 hour period to obtain the net daily change in DO concentrations. During periods of low solar radiation and re-aeration we can expect this to produce a declining level of DO. Hypoxic conditions are produced when there is a sequence of such days.

Model Calibration

The model was calibrated using the observed depth-averaged dissolved oxygen concentrations. Calibration data was available for all model variables for two periods 15 November to 4 December in 2012 (Figure 8.4A) and 22 January to 7 March in 2013 (Figure 8.4B). The model parameters were calibrated using three hourly mean oxygen concentrations. Calibrated coefficients and exponents used in this DO model are shown in Table 8.1.

Table 8.1: Parameters calibrated for the Dissolved Oxygen Model

P_{max}	6.85 $gO_2m^{-2}s^{-1}$	K_R	0.003 s^{-1}	b_1	1.13
K_P	17.17 Wm^{-2}	a_1	0.20	b_2	1.12
k_1	0.0015	a_2	0.14	b_3	1.11
E	3.76 $gO_2m^{-2}s^{-1}$	a_3	3.15	k_3	0.017
Q	0.017				

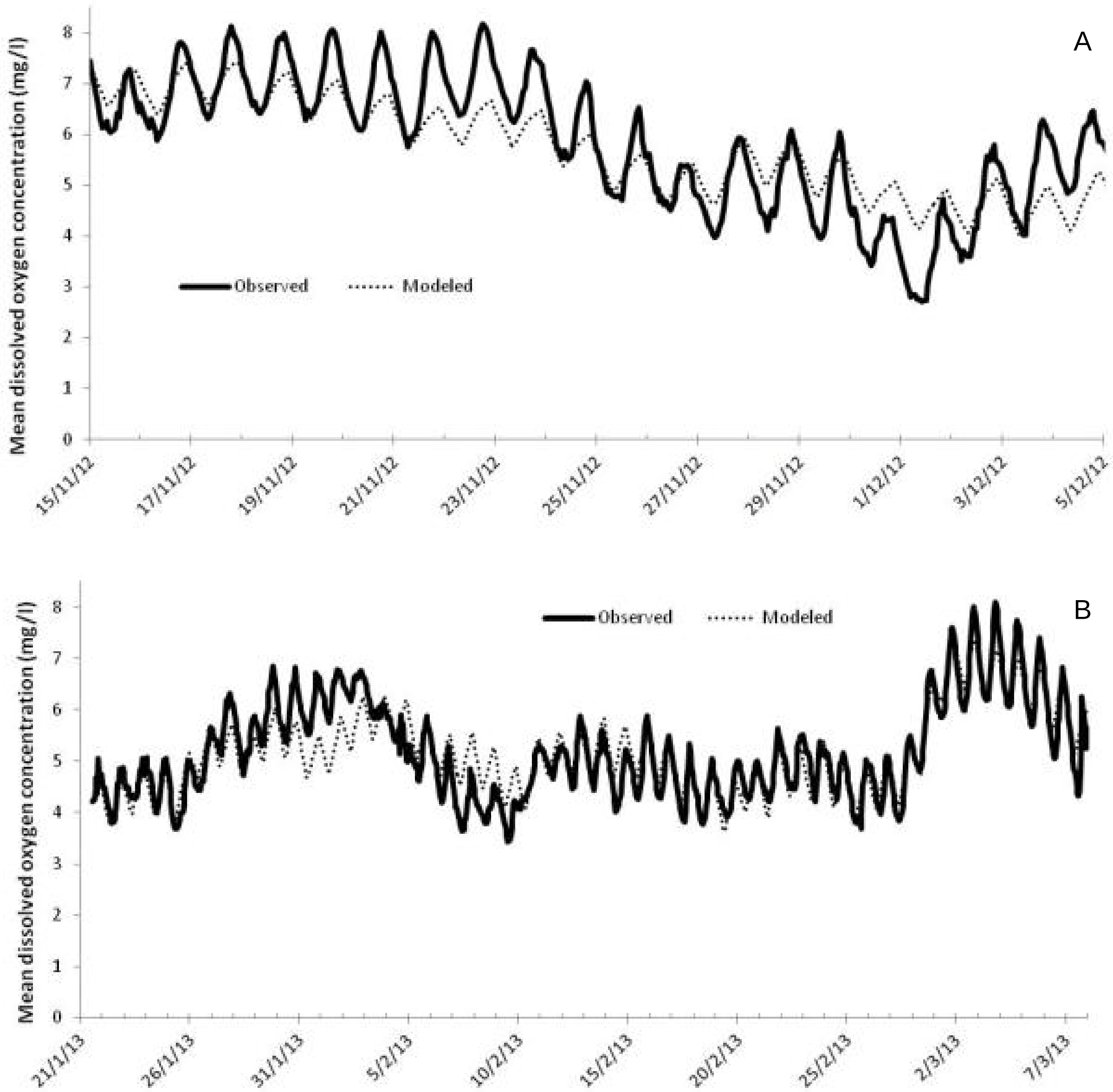


Figure 8.4: Modeled and observed time series of dissolved oxygen concentrations for A) late November 2012, and B) January through early March 2013

Whilst the model performed well in reproducing diurnal variation in dissolved oxygen (Figure 8.4), it is more critical that daily net changes in DO are reproduced. It is a net reduction in DO over a sequence of days that will lead to hypoxic conditions. Figure 8.5 shows the modeled daily net change in DO concentration plotted against the observed net change. This is the critical aspect of model performance rather than sub-daily changes in DO. Model performance is good given the assumptions required in model formulation.

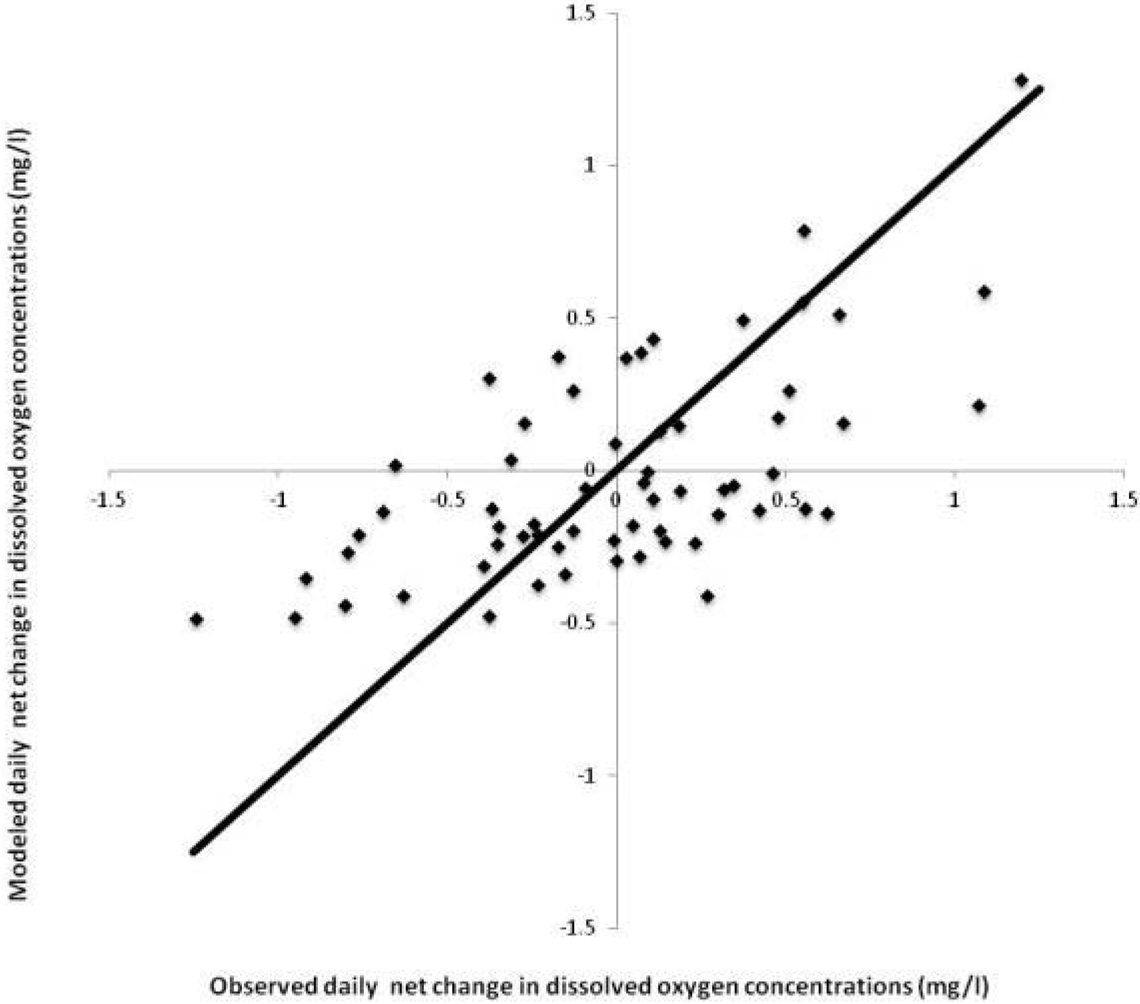


Figure 8.5: Modeled and observed daily net change in dissolved oxygen concentrations (line indicates 1:1 ratio)

Mixing criterion - R. Bormans and Webster (1997) developed a ratio that links the relative amount of stratifying surface heating with destratifying mixing induced by flow. This criteria – *R* – differs from the Richardson number in that it is a model of the fluxes associated with heating, cooling and mixing of the water column, rather than a description of the gradients that have already formed. Consequently, the parameter *R* can be used to estimate the flow requirements with which to prevent the formation of stratified conditions (Whiterod and Sherman, 2012). *R* is given by equation 6, where Q_{net} is the net heat flux into the river, Q_I is the net short wave radiation, C_p is the specific heat capacity of water, α is the thermal expansion coefficient and K_d is the light attenuation coefficient. Detailed calculation methods are available in Bormans and Webster (1997).

6. $$R = \frac{U^3}{L\left(Q_{net} - \frac{2Q_I}{K_d L}\right)\frac{\alpha g}{\rho C_p}}$$

This mixing criterion accounts for seasonal effects on solar energy input, and adjusts the daily energy input according to the hour of the day. However, it is not valid when the water column is cooling, so analysis has been limited to daylight hours. To assess the flow required in Broken Creek to prevent temperature stratification, *R* was calculated hourly using data for water temperature, solar radiation, air temperature, relative humidity, wind speed, and light attenuation. Data was output at 5 time points, 6:00 am, 9:00 am, 12 noon, 3:00 pm and 6:00 pm. Fixed flow rates were incrementally increased to establish the flow rate required to achieve complete mixing of the water column throughout the diurnal cycle and these daily results were averaged over the summer months (December – February).

8.4 Results

Between October 2012 and March 2013, environmental water made up the vast majority of flow delivered through Rices Weir. In early October 2012, there was high flow through Rices Weir that was not sourced from environmental flows, but between November 2012 and February 2013, 93% of flow constituted environmental water.

Azolla cover was estimated at less than 5% for the intensive data collection period (October 2012 – March 2013). Consequently, this was considered to have negligible impact on surface re-aeration and light irradiance at Rices Weir for modeling purposes. However, on 12 May 2013, Azolla growth had extended significantly (~15 m) from the shore and was accumulating around the upstream opening to the fish ladder (Figure 8.6).

The water column at Rices Weir was consistently well-mixed, with temperature, dissolved oxygen and water velocity profiles showing little persistent change with depth (Figure 8.7 & Figure 8.8). Throughout the 2012-13 summer, the calculated Richardson number was consistently at or below zero, and temperature stratification was either non-existent or weak and transient. This suggests that flow-induced turbulence was sufficient to overcome any buoyancy formation from incoming solar energy into the surface water of the weir pool. The persistence of a mixed water column also justifies the assumption of uniform DO concentrations with depth and the use of a 1-dimensional model to represent the daily and diurnal patterns in DO. In addition, persistent well-mixed conditions suggest that changes in

DO result from the net effect of oxygen production, consumption and re-aeration across the full water column. The *R* criterion, averaged for the summer season, suggested that transient stratification at the peak solar height and intensity (around 1300 h) was likely to occur at discharge volumes below 500 ML/day (Figure 8.10). However, at flows in line with those observed during the 2012-13 summer (250 – 300 ML/day), any such stratification would have been highly unstable as the energy input associated with the lower and cooler morning and afternoon sun was insufficient to overcome the turbulence associated with environmental flows. Overnight cooling of the water surface would further reduce the chance of stratification persisting over a 24 hour period.

The DO balance model shows that the rate of change in water column DO from photosynthesis ranges from 0 $mg.l^{-1}.hr^{-1}$ overnight to 0.196 $mg.l^{-1}.hr^{-1}$ during maximum solar radiation and low turbidity levels. The oxygen demand for respiration and loss to the sediment ranged between 0.16 $mg.l^{-1}.hr^{-1}$ to 0.12 $mg.l^{-1}.hr^{-1}$, with higher values related to warmer water temperatures. Exchange of oxygen from the atmosphere to the water column was dependent on the level of oxygen saturation in the water column, wind speed and direction, and the water velocity. This rate of exchange ranged from 0.107 $mg.l^{-1}.hr^{-1}$ to -0.007 $mg.l^{-1}.hr^{-1}$. The (downward) exchange of oxygen between the water and the atmosphere was positive for all days modeled except for a few hours on 5 December 2012, which suggested that the atmosphere was a source of oxygen to the water column, and that this re-aeration increased with environmental flows.

Figure 8.6: Azolla, which was not prevalent between October 2012 and March 2013 (e.g. left panel taken on 17 January 2013), extended about 15 m from the shore near the fish ladder at Rices Weir on the 12 May 2013 (right panel)

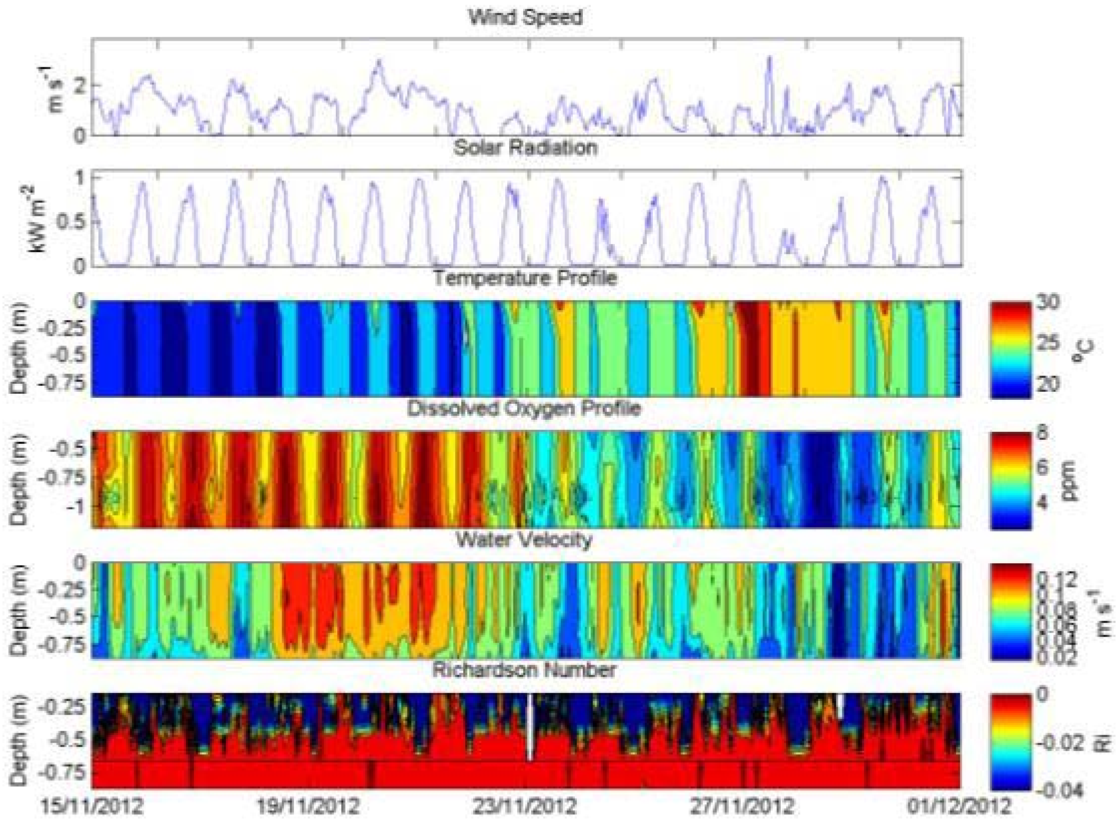


Figure 8.7: Physical parameters and calculated discrete Richardson Number for Rices Weir between 15 November and 5 December 2012

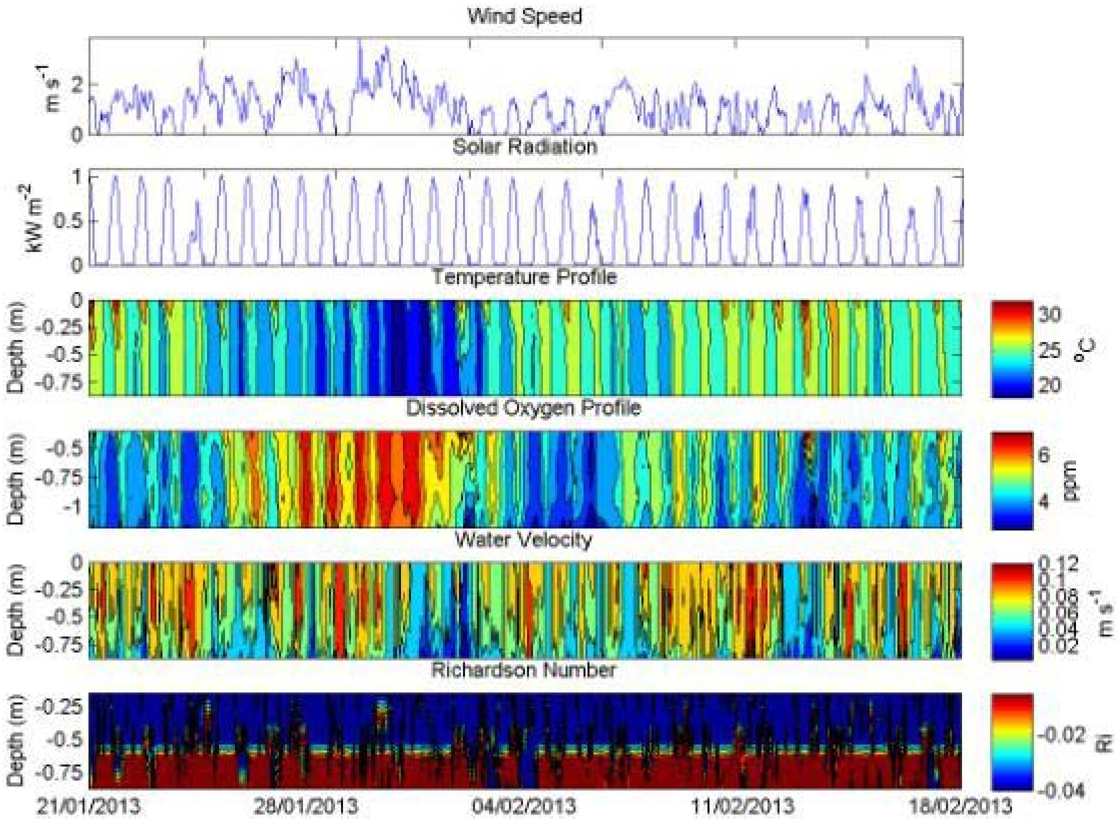


Figure 8.8: Physical parameters and calculated discrete Richardson Number for Rices Weir between 21 January and 18 February 2013.

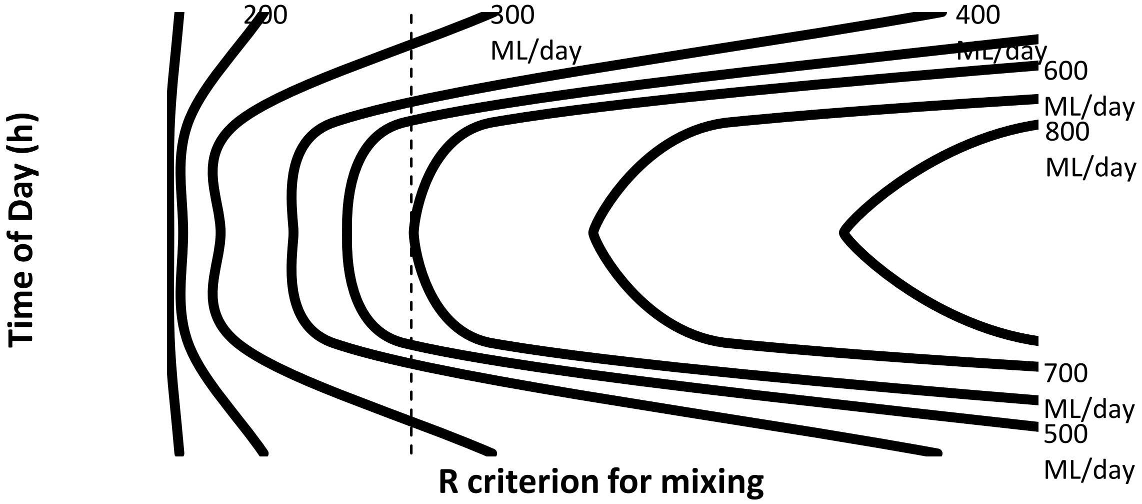


Figure 8.9: The R criterion for mixing shows the relationship between different daily flow velocity (contours) with time of day. The transition from a mixed to a stratified water column occurs as R falls below approximately 45000 (dotted vertical line).

The effectiveness of environmental flow delivery can be simulated over the periods that the DO balance model was calibrated by manipulating the flow input into the model. This allows a comparison between the observed DO conditions and that of a hypothetical or counterfactual condition where no environmental flow is delivered. Without environmental flows, depth-averaged water column DO concentrations decrease to zero over both periods of simulation. In November 2012, simulated DO concentrations fell to zero after 11 days (Figure 8.10A), suggesting that photosynthesis and re-aeration without flow were not sufficient to offset high respiration and sediment oxygen demands. Similarly, between the 21 January and 7 March 2013, modeled DO concentrations reached zero on the 1 February (Figure 8.10B). Modeled DO concentrations remained low in both simulations for the duration of the simulation with only brief periods where the minimum DO concentration during each diel cycle did not reach zero. As the rate of photosynthesis was not a function of flow in this model, the declining DO concentrations resulted from the accumulated daily oxygen deficit caused by a decreased level of turbulence-driven exchange with the atmosphere.

Data from two 48 hour periods where compared to examine the major differences between the highest and lowest flows measured during the study period. These results are shown in

Table 8.2 and show that, as expected, the temperature difference between the surface and bottom of the water column was higher during the period of low flow compared with the period of high flow. This occurred despite higher solar energy during the period of high flow. However, the oxygen gradient during the high flow period was higher than that during the low flow period, which potentially reflected the flow-induced aeration as shown by the greater surface oxygen concentrations.

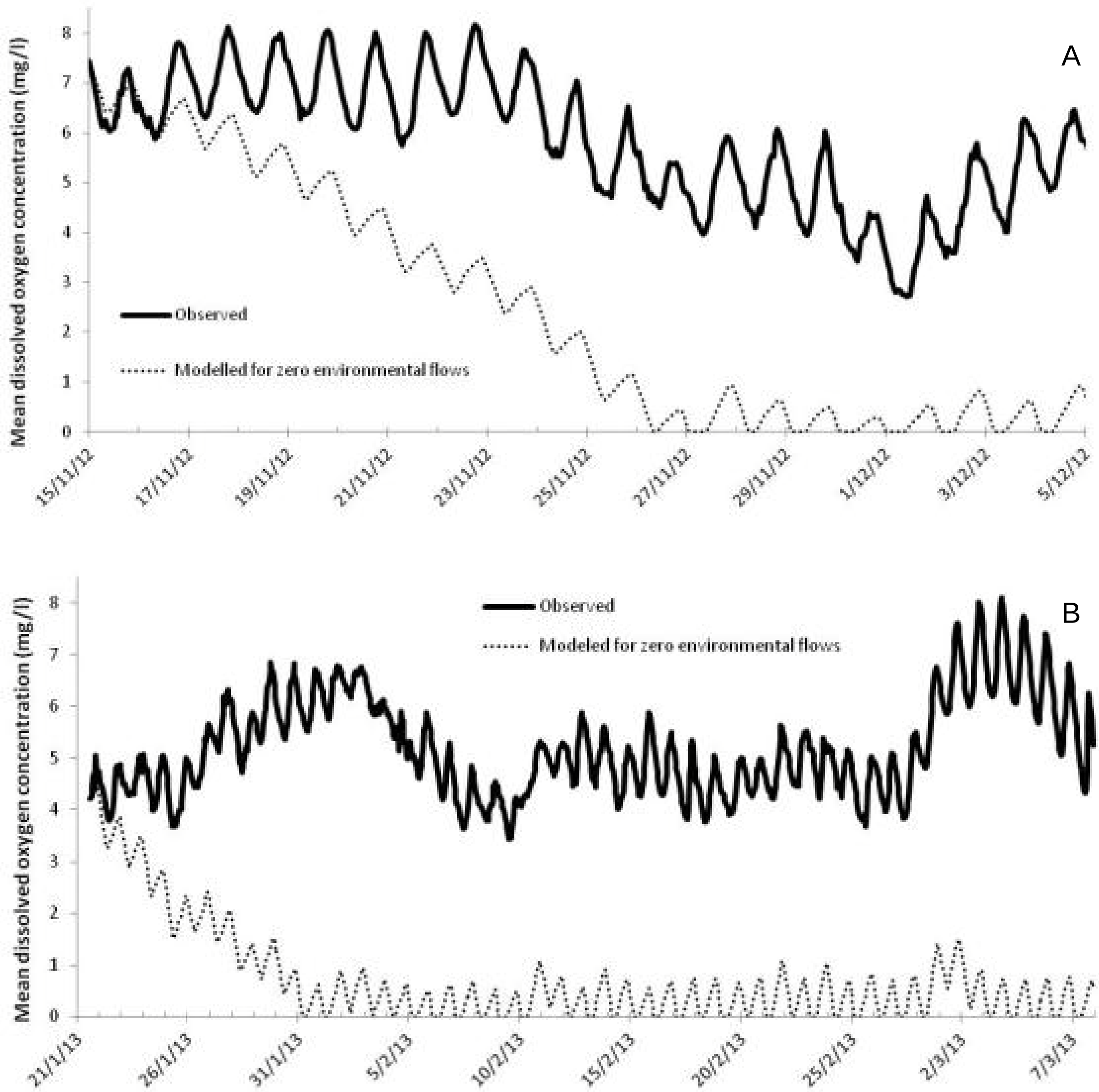


Figure 8.10: Comparison of observed DO concentrations with those modeled without environmental flow (average discharge reduced from 332.7 ML/day to 23.3 ML/day)

Table 8.2: Comparison of two periods of 48 hours each with different flow volumes

Date	28/2 – 1/3	17/3 – 18/3
Condition	**High Flow**	**Low Flow**
Average *U*	466.7 ML/day	92.7 ML/day
Average Temp	23.1°C	19.5°C
Max Temp Difference	0.67°C	2.22°C
Max DO Difference	3.59 mg/L	1.32 mg/L
Average Surface DO	5.9 mg/L	5.1 mg/L
Minimum DO	2.8 mg/L	4.1 mg/L

8.5 Discussion

Commonwealth environmental water made up the majority of flow through the lower Broken Creek between October 2012 and March 2013. The available evidence suggests that without this environmental flow, water quality, in terms of DO concentrations, in Rices Weir pool would have been substantially affected with extended periods of hypoxia. In the absence of the environmental flow, modeled results indicate that DO concentrations would have undergone rapid and sustained depletion of DO, well below the ANZECC water quality guidelines.

The 1-dimensional model used for this assessment requires a number of assumptions and its output should be considered with caution. Importantly, the model could only be calibrated during periods of sustained flow with a well-mixed water column so predictions under low or no flow are uncertain.

As flow declines, an additional level of complexity is added to interactions between flow and dissolved oxygen levels as stagnant waters become thermally stratified. Extensive stratification was observed in previous studies of Rices Weir, with temperature differences between the top and bottom of the water column measured as high as 9 degrees in 2006 (Rees *et al*. 2007). Under these conditions, steep density gradient between the two layers of water prevent solute from mixing through the water column, and DO transfer is severely limited across this gradient into the lower layer of colder, denser water called the hypolimnion (Turner & Erskine 2005). In this way, stratification changes the relative influence of different processes in the bottom and surface waters. Sediment mineralisation rates influence the smaller volume of water below the temperature gradient, photosynthetic production of oxygen is generally confined to the upper water layer where light is sufficient, and atmospheric exchange replenishes only the surface layers of water with oxygen.

Consequently, deeper water becomes more rapidly depleted of oxygen, while the upper layer remains replete with DO.

A flow manipulation event had been planned for January 2013 where flow was to be reduced substantially in order to provide the necessary conditions for temperature stratification to form and provide data to develop the DO model. However, challenging and unpredictable river-operating conditions during the hot weather made river operations difficult and this flow manipulation could not be delivered (G. Earl, 2013, pers. comm.).

Environmental flows have been very effective in preventing any substantial stratification as the Richardson number remained at or below zero. This indicated that flow provisions sustained water velocity above a sufficient magnitude to overcome buoyancy formation. Temperature stratification was therefore only ever transient (lasting less than 4 hours) during this experiment.

This study provides strong evidence that environmental flows have been effective in maintained moderate to high DO concentrations. Further, model results show that in the absence of environmental flows, low DO concentrations would have persisted throughout the period of monitoring.

The positive effect of environmental flows is produced through a number of mechanisms. As has already been indicated, flow maintains a mixed water column that prevents the formation of an anoxic hypolimnion and it generates turbulence at the water surface promoting oxygen exchange with the atmosphere (Stefan & Fang 1993). Increased flow is often induced by greater rainfall and surface run-off, which can flush suspended solids from the riparian zone into the stream. Flow also induces turbulence that can maintain higher concentrations of entrained sediments, or can scour sediments from the stream bed. Consequently, higher flows are often coincident with increased turbidity levels that reduce light penetration and photosynthetic production of DO (Vervuren *et al*. 2003; Nilsson & Renöfält 2008), although this effect is not evident in data collected during this study. Flow-induced mixing also supports transitioning of phytoplankton type from buoyant cyanobacteria towards those reliant on mixing, such as diatoms (Sherman *et al*. 1998).

While the counterfactual model simulation of conditions without environmental flows suggests that the reduction in re-aeration dominated the decline in DO concentrations, low levels of photosynthetic production or relatively high levels of respiration and sediment oxygen demand could also have exacerbated this effect. The highly turbid water during the experimental period may have reduced light availability and inhibited photosynthetic production of DO, however this is unlikely to affect the model output significantly. It is evident that turbidity measured between November 2012 and March 2013 (above 225 NTU) was significantly higher than previous studies at Rices Weir (130 NTU, Rees *et al*. 2007) and elsewhere in the Murray-Darling Basin including the Goulburn River (above 50 NTU, Fletcher *et al*. 1985), the River Murray above the confluence with the Darling River (~28 NTU, Walker & Thoms 1993) and Lake Alexandrina (80 – 120 NTU, Skinner 2011). The turbidity levels were also significantly higher than the period either side of the study presented here. However, modeled estimates of photosynthesis in Rices Weir reached 4.704 $g.m^{-3}.d^{-1}$, which is consistent with estimates from the literature, with Rees *et al*. (2007) measuring production rates between 1 – 3 $g.m^{-3}.d^{-1}$ in Broken Creek, and Oliver and Merrick (2006) reporting summer oxygen production rates of 4.2 $g.m^{-3}.d^{-1}$ for a site on the River Murray in the Barmah Forest, in the vicinity of Rices Weir.

As the model treated the entire water column as a single, uniform cell, both sediment oxygen demand and community respiration rates were added together to give a total oxygen demand of 3.1 $g.m^{-3}.d^{-1}$. This is comparable, but at the lower end of previously measured summer oxygen demand in the Broken Creek that ranged between 2.3 – 12.9 $g.m^{-3}.d^{-1}$ (Rees *et al*. 2007). This lower value used in modeling is consistent with oxygen consumption rates measured during this study. Any underestimation of the total oxygen demand would also manifest as higher modeled oxygen concentrations, suggesting that appropriate rates of oxygen consumption were input into the model.

The average re-aeration component used in the oxygen balance model of 0.51 $m.day^{-1}$ was consistent with values used in the literature. Stefan and Fang (1993) reported surface oxygen transfer velocity between 0.5 – 1 $m.d^{-1}$ for average wind speeds over seven lakes. For their experimental site at Barmah, in the River Murray, Oliver and Merrick (2006) reported an average re-aeration component of 0.61 $m.d^{-1}$ and a maximum of 1.16 $m.d^{-1}$ in 1999. The maximum re-aeration component used at Albury, on the River Murray, upstream of Barmah where water velocities are higher, was 3.19 $m.d^{-1}$ (Oliver & Merrick 2006).

We have shown that a one-dimensional, deterministic dissolved oxygen model successfully simulates the diurnal and monthly changes to DO. A simulation of changes to DO in the lower Broken Creek without Commonwealth environmental water provisions shows a rapid and sustained reduction in dissolved oxygen concentrations, highlighting the importance of this environmental water delivery. Flow-induced mixing also maintained a thermally uniform water column, with only transient periods of weak stratification. The combination of flow, well-mixed water and more stable dissolved oxygen concentrations also appeared to prevent any development of Azolla blooms. Azolla may have been further minimised because the weir pool is dominated by surface flow, which would reduce the residence time of the floating fern. In mid-May 2013, some Azolla growth was once again visible. More sophisticated three-dimensional modeling, which couples the dissolved oxygen balance presented here with a heat transport model to consider energy budgets (e.g. Bormans & Webster 1998), could be considered once field data to calibrate the link between flow and the formation and erosion of temperature stratification is available. This coupled model would provide a better insight into the timing and duration of flow requirements to maintain DO conditions at Rices Weir and in the lower Broken Creek.

9 DISSOLVED OXYGEN AND FISH

9.1 Evaluation question and expected outcomes

This section addresses the evaluation question: How is fish movement or activity affected by changes in dissolved oxygen and flows in the lower Broken Creek? The expected outcome was that fish would avoid water that is low in dissolved oxygen (DO).

9.2 Methodology

Background

Hypoxia (low dissolved oxygen) is a serious water quality issue in the Murray-Darling Basin that has the potential to inflict major adverse impacts upon native fish populations (Beesley et al., 2013). The Broken Creek in northern Victoria has a diverse native fish population, which includes several species of recreational angling value and/or conservation significance, in particular Murray Cod (*Maccullochella peelii*) and Golden Perch (*Macquaria ambigua*). Major fish kills have become a serious problem in lower Broken Creek in recent years (Koehn, 2005, Rees, 2006). The most likely explanation for the fish kills is low DO concentrations (hypoxia) in conjunction with little or no flow (Koehn, 2005, Rees, 2006). This study aims to examine relationships between low DO events, flows and the behaviors of native fish in Broken Creek using acoustic telemetry. In particular, the approach aims to determine how fish respond to low DO in terms of their levels of activity (i.e. do fish shut down physiologically?), their position in the water column (i.e. how do fish respond to vertical DO stratification?), and whether fish move away from reaches affected by low DO.

Acoustic telemetry

Murray cod (*n* = 23, median total length [TL] 410 mm, range 320-780 mm) and Golden Perch (*n* = 15, median TL 348 mm, range 315-415 mm) were collected from Broken Creek between 0.1 and 4 km upstream of Rices Weir, by using a boat-electrofishing unit in August 2012 (Table 9.1, Figure 9.1, Figure 9.2). This reach was selected for the study because low dissolved oxygen and fish deaths have been identified as a concern in the lower sections of the creek immediately upstream of Rices Weir (Rees, 2006). On capture, fish were immediately transferred into an aerated 50-L holding container and individually anaesthetised (0.03 ml AQUI-S L^{-1} water) (AQUI-S, Lower Hutt, New Zealand). Two types of 69-kHz transmitters (Vemco, Nova Scotia, Canada; average delay between transmissions: 250

seconds) were used depending on the size of the fish. For smaller fish (< 800 g), V9AP transmitters (dimensions: 46 x 9 mm; weight: 6.3 g in air) were used. For larger fish (> 800 g), V9AP or V13AP transmitters (dimensions: 44 x 13 mm; weight: 12.3 g in air) were used. These transmitters measure acceleration (activity) and depth of tagged fish as they move within an acoustic receiver array. The acceleration signal is measured in terms of m s^{-2} (SI units) and it is a vector quantity that is a result of measuring acceleration on 3 axes (X,Y,Z) (Vemco.com). The transmitters were implanted into the peritoneal cavity through an incision of 12–14 mm, on the ventral surface between the pelvic and anal fins. Up to 3 interrupted external sutures were used to close the incision. Only fish over 400 grams were tagged ensuring that the transmitter to fish weight ratios remained below about 2% (Winter, 1996). Larger fish were also tagged with PIT tags (passive integrated transponder) to provide information on movement through fishways where PIT tag readers are installed. Each fish was placed into a recovery net positioned in the stream channel. Once the fish were observed to maintain their balance and freely swim throughout the holding net they were released near their point of capture (Figure 9.3 and Figure 9.4).

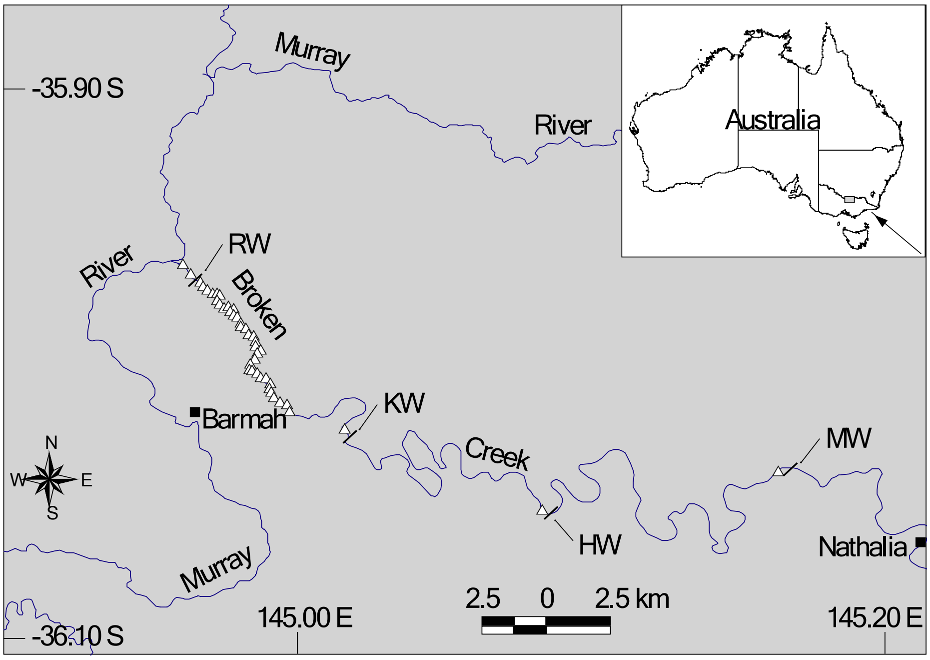


Figure 9.1: Map showing location of the study site. White triangles represent the locations of each of the listening stations in Broken Creek. RW - Rices Weir, KW - Kennedys Weir, HW - Hardings Weir, MW - Magnussons Weir

Table 9.1 Summary of tagging details for Murray Cod and Golden Perch collected in Broken Creek.

Species	Fish ID	Length (mm)	Weight (grams)	Tag type
Murray Cod	M1	320	410	V9
Murray Cod	M2	320	422	V9
Murray Cod	M3	325	440	V9
Murray Cod	M4	350	536	V9
Murray Cod	M5	344	559	V9
Murray Cod	M6	360	645	V9
Murray Cod	M7	366	692	V9
Murray Cod	M8	390	799	V9
Murray Cod	M9	404	967	V9
Murray Cod	M10	425	1122	V9
Murray Cod	M11	407	1162	V9
Murray Cod	M12	470	1807	V9
Murray Cod	M13	698	6381	V9
Murray Cod	M14	702	6785	V9
Murray Cod	M15	390	876	V13
Murray Cod	M16	410	975	V13
Murray Cod	M17	440	1111	V13
Murray Cod	M18	510	2022	V13
Murray Cod	M19	550	2688	V13
Murray Cod	M20	650	5184	V13
Murray Cod	M21	710	6255	V13
Murray Cod	M22	710	7083	V13
Murray Cod	M23	780	10000	V13
Golden Perch	G1	314	523	V9
Golden Perch	G2	321	565	V9
Golden Perch	G3	320	585	V9
Golden Perch	G4	330	636	V9
Golden Perch	G5	340	664	V9
Golden Perch	G6	348	702	V9
Golden Perch	G7	340	719	V9
Golden Perch	G8	340	730	V9
Golden Perch	G9	360	782	V9
Golden Perch	G10	357	802	V9
Golden Perch	G11	350	808	V9
Golden Perch	G12	354	855	V9
Golden Perch	G13	379	994	V9
Golden Perch	G14	400	1228	V9
Golden Perch	G14	415	1408	V13

Figure 9.2: Collecting fish in Broken Creek using the boat electrofishing unit

Figure 9.3: Murray cod being released after tagging in Broken Creek

Figure 9.4: Golden perch being released after tagging in Broken Creek

Forty-two acoustic listening stations (Model VR2W, Vemco) were deployed in August 2012 in Broken Creek between the River Murray junction and Magnusson's Weir near Nathalia (a distance of about 50 km) (Figure 9.1, Figure 9.5). Thirty seven of these listening stations were deployed between Rices Weir and 7 km upstream at about 200 m intervals to provide more precise information on fish locations within these reaches. Range tests showed that the listening stations had detection ranges of about 100 m depending on the physical attributes of the site (e.g. depth) and therefore the array of listening stations provides almost continuous spatial coverage of fish movements in this reach. The listening stations were deployed using a length of plastic-coated steel cable attached to trees as anchor points. Data were downloaded from the listening stations monthly for the duration of the study.

Figure 9.5: Acoustic listening station being deployed in Broken Creek

Environmental variables

Daily discharge, water temperature and dissolved oxygen (surface and bottom) records were obtained from a gauging station at Rices Weir. Water temperature and DO were also measured using a data logger (D-Opto, Zebra Tech, Nelson, New Zealand) (surface and bottom) at 1.5, 3.0 and 4.5 km upstream of Rices Weir, and using a handheld water quality meter (90FL-T, TPS, Brisbane, Australia) every 600 m (surface and bottom) between 0.1 and 6 km upstream of Rices Weir fortnightly between October 2012 and February 2013.

Data analysis

Bayesian analyses were used to examine relationships between the median hourly depth and activity of Murray Cod and Golden Perch and environmental variables (Kéry, 2010). The environmental variables were, mean hourly flow, change in hourly flow, mean hourly dissolved oxygen, change in hourly dissolved oxygen, mean hourly water temperature and previous hourly activity or depth. For activity, a quadratic term for temperature was also included to account for possible peak activity suggested by the data. The form of the model is given below:

Variables	*Definition*
μ_t	Expected activity/depth at time *t*.
σ^2	Variance of activity/depth.
λ	Expected activity of fish when environmental variable are average.
σ_{Fish}^2	Between fish variance.
Response	
Activity	Median hourly fish activity (m s^{-2}).
Depth	Median hourly fish depth (m).
Explanatory	
Flow	Mean hourly flow (standardised).
ΔFlow	Difference in current mean hourly flow compared to previous hours mean flow.
DO2	Mean hourly dissolved oxygen (standardised).
ΔDO2	Difference in current DO2 compared to previous DO2.
Temp	Mean hourly temperature (standardised).
ε_j	Random intercept associated to fish *j*.

$$Activity_t \sim N(\mu_t, \sigma^2)$$

$$\mu_{t+1} = \beta_1 Activity_t + \beta_2 Temp + \beta_3 Temp^2 + \beta_4 DO + \beta_5 \Delta DO + \beta_6 Flow + \beta_7 \Delta Flow + \varepsilon_j$$

$$\varepsilon_j \sim N\left(\lambda, \sigma_{Fish}^2\right)$$

The models were constructed in JAGS (Plummer, 2003) via R (R Development Core Team, 2013) using the package R2jags (Su and Yajima, 2012). Model chains were run until the chains converged. Convergence was defined as having all Gelman and Rubin's convergence diagnostic potential scale reduction factors being less than 1.05 (Gelman et al, 2004). If the 95% high-density interval (HDI) did not include zero, the parameter was considered to influence activity and depth. Assessing the model fit was done using posterior predictive Bayesian *p*-values and Pearson residuals. A posterior predictive Bayesian *p*-value close to 0 or 1 is evidence that the model does not fit the data well (Gelman et al., 1996). Similarly Pearson residuals greater than 4 may indicate poor fit. The analyses excluded fish data collected before October because fish may exhibit altered activity or behaviour immediately after tagging (Hilderbrand and Kershner, 2000, Koster and Crook, 2008).

9.3 Results

Hydrology, dissolved oxygen and temperature

Flow was highest around October before receding to lower levels through to February (**Figure 9.6**). Environmental water was delivered in order to provide a minimum of 250 ML d^{-1} at Rices Weir from 1 September to 29 January. The aim was to increase native fish habitat and allow dispersal of native fish throughout Broken Creek (GBCMA, 2012). A minimum of 200 ML d^{-1} was then delivered from 30 January to 15 May, with the aim of maintaining DO concentrations above 5 mg L^{-1} (GBCMA, 2012). During this period, CEWH water comprised: 100 ML d^{-1} from 1 September to 5 October, 250 ML d^{-1} to 12 December, 160 ML d^{-1} to 31 December, 250 ML d^{-1} to 29 January, and 200 ML d^{-1} from 30 January to 15 May. Several increased flow events driven by rainfall runoff from the catchment also occurred during the study (e.g. from 184 to 449 in September ML d^{-1}, 190 to 705 ML d^{-1} in October). The use of water for irrigation also resulted in flow variations at times (e.g. from 251 to 174 ML d^{-1} in December) (Geoff Earl, pers. comm.).

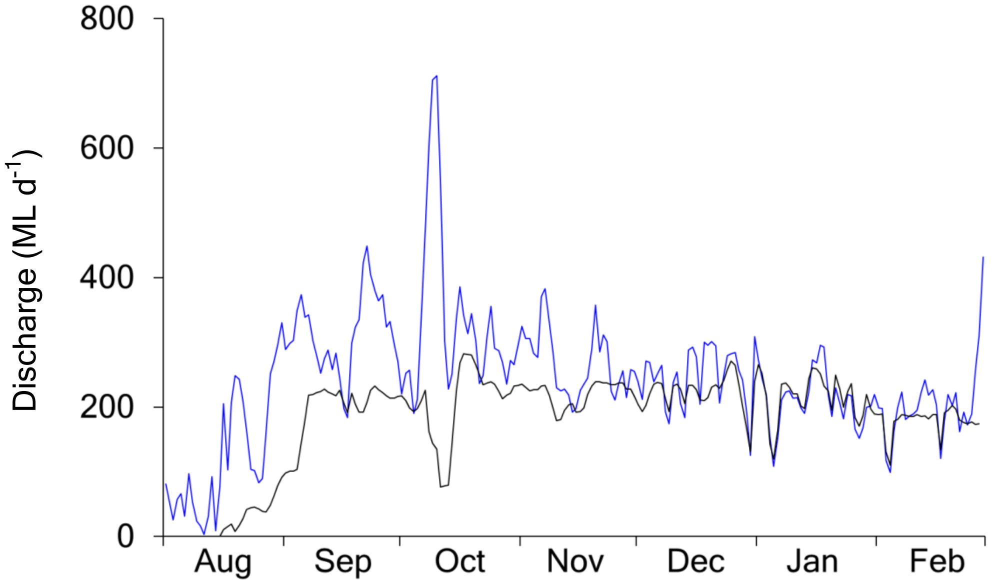


Figure 9.6: Discharge (blue line) in Broken Creek at Rices Weir. Grey line represents total environmental water released.

Logged readings of DO and water temperature showed similar trends across the three sites between 1.5 and 4.5 km upstream of Rices Weir over the study period (Figure 9.7, Figure 9.8). Spot measurements of DO and temperature were also similar across sites between 0.1 and 6.6 km upstream of Rices Weir (Figure 9.9). Logged readings of DO generally remained above 4 mg L^{-1} over the study period. However, there were short periods (i.e. < 1-2 d duration) when DO decreased to about 2.5-3 mg L^{-1} in early December, early January and early February. DO at the bottom (mean 6.38 ± 0.01 mg L^{-1} SE) and surface (mean 6.31 ± 0.01 mg L^{-1} SE) were generally similar, although there were occasional short periods (i.e. < 1 d duration) when bottom DO was about 2-3 mg L^{-1} lower than surface. Maximum temperatures of around 33° C were reached in January and minimum temperatures of around 13° C occurred in September. Water temperatures at the bottom (mean 22.96 ± 0.02° C SE) and surface (mean 22.55 ± 0.03° C SE) were generally similar.

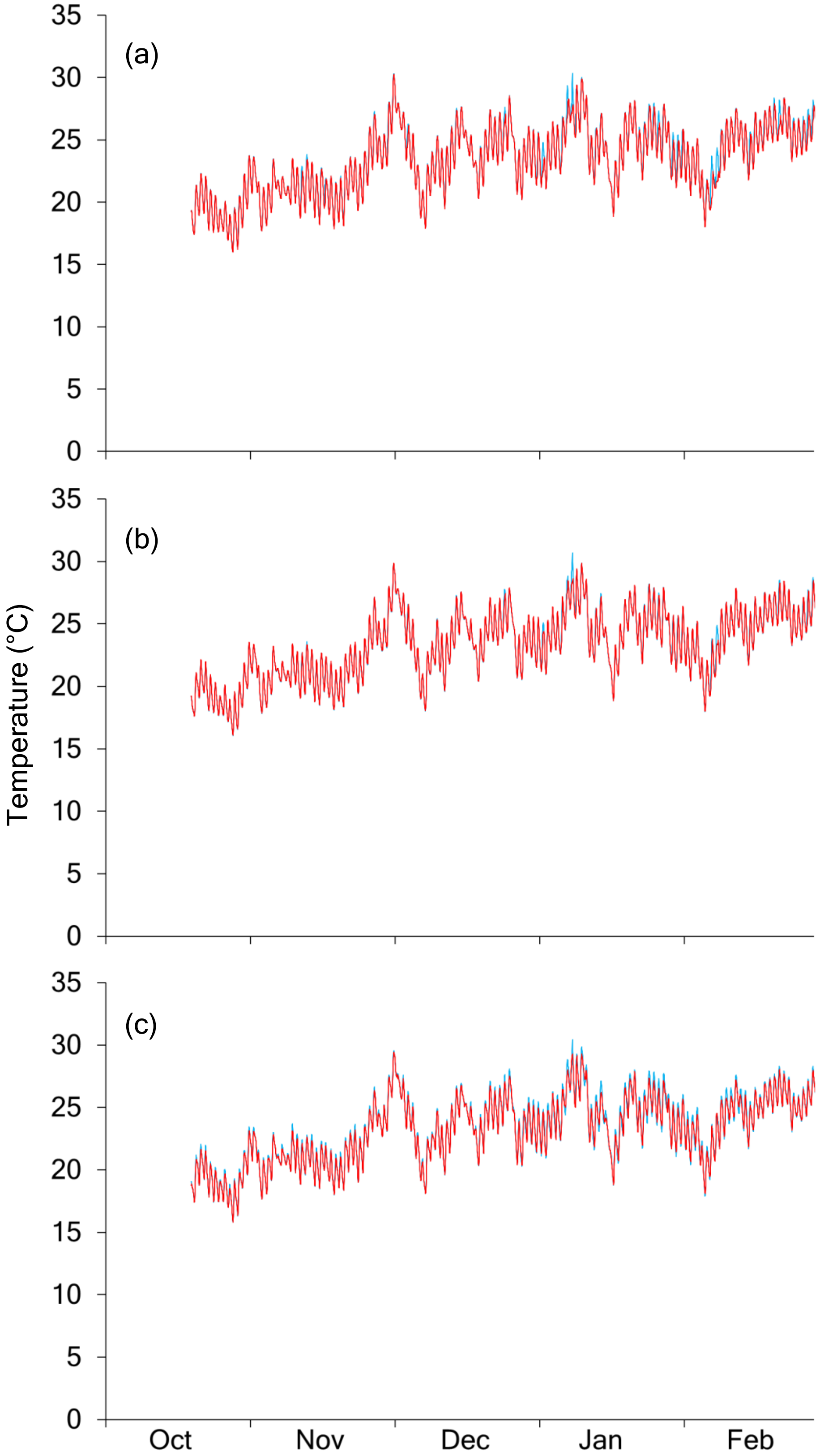


Figure 9.7: Bottom (red line) and surface (blue line) water temperature in Broken Creek at (a) 1.5, (b) 3.0 and (c) 4.5 km upstream of Rices Weir.

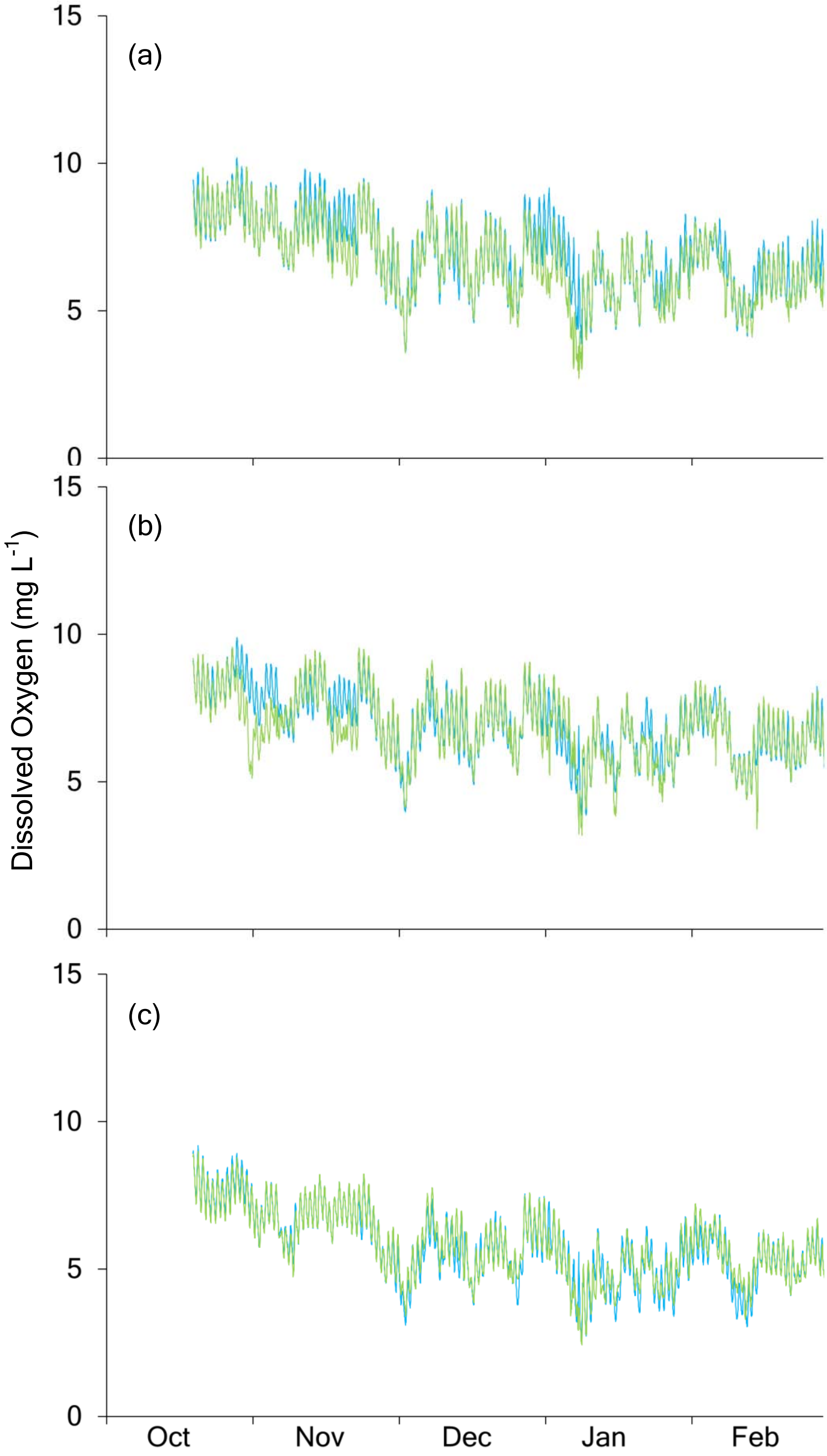


Figure 9.8: Bottom (green line) and surface (blue line) dissolved oxygen in Broken Creek at (a) 1.5, (b) 3.0 and (c) 4.5 km upstream of Rices Weir.

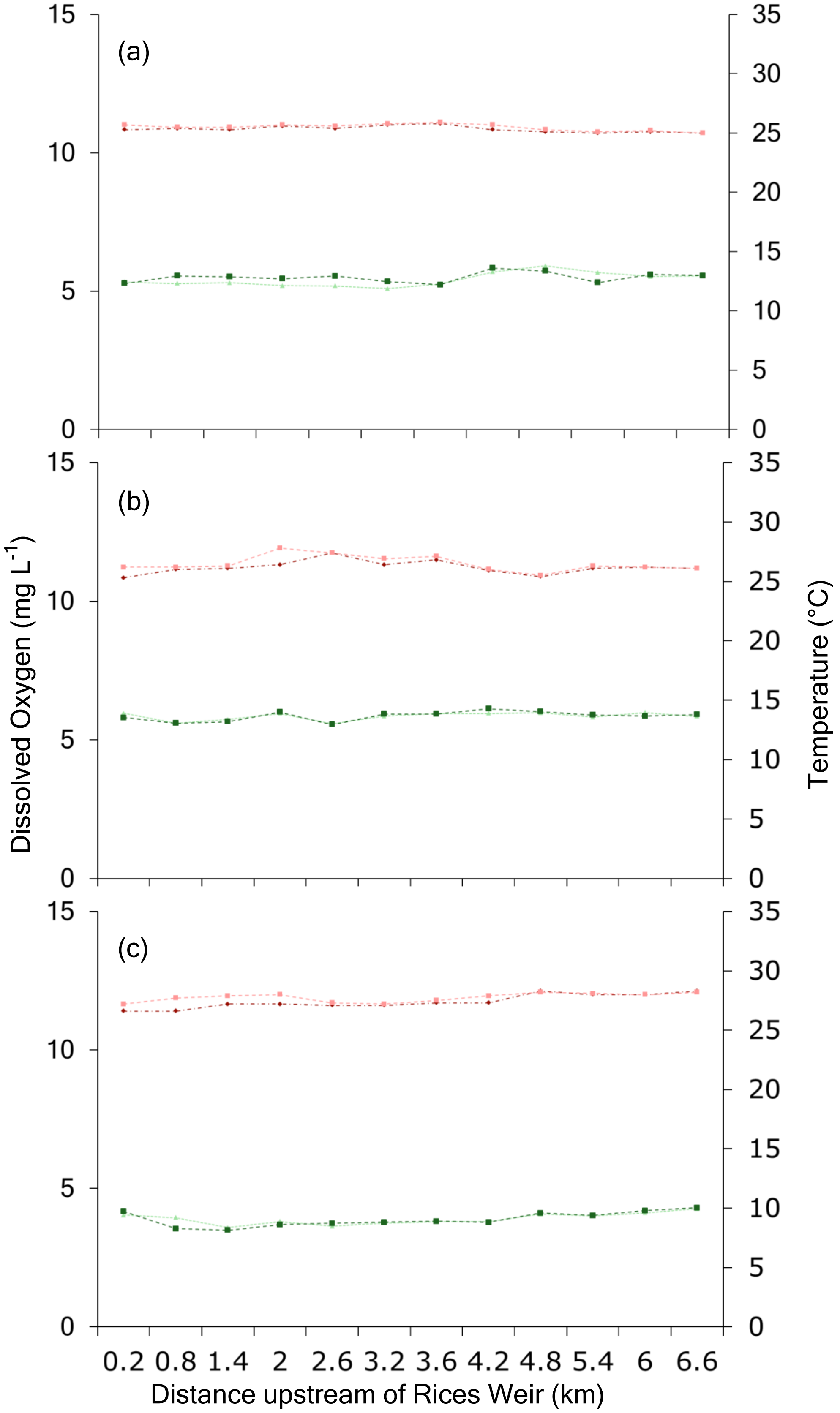


Figure 9.9: Examples of spot measurements of dissolved oxygen (surface - light green line, bottom - dark green line) and temperature (surface - light red line, bottom - dark red line) upstream of Rices Weir on (a) 26/11/2012, (b) 12/12/2012 and (c) 7/1/2013.

Acoustic tracking

Activity and depth of Murray Cod and Golden Perch

Murray Cod and Golden Perch activity was relatively low for about 4 weeks after tagging, likely as a result of post-surgery recovery (Figure 9.10 and Figure 9.11). Murray Cod and Golden Perch activity ranged from 0.0-4.9 m s^{-2} (median 0.57 m s^{-2}) and 0.1-3.2 m s^{-2} (median 0.61 m s^{-2}), respectively. For both species activity values were similar at night (Murray Cod: median 0.57 m s^{-2}, Golden Perch: median 0.65 m s^{-2}) compared to the day (Murray Cod: median 0.55 m/s^2, Golden Perch: median 0.59 m s^{-2}) (Figure 9.10 and Figure 9.11).

Murray Cod and Golden Perch activity tended to gradually increase from August through to a peak in November-December and then decline or plateau to February (Figure 9.10 and Figure 9.11). Murray Cod activity exhibited several pronounced spikes in October, coinciding with sharp rises in temperature (e.g. from 16.2 to 23.2° C in late October) (Figure 9.10). However, activity in neither species appeared to be strongly influenced by any of the environmental variables examined when compared to the unexplained variation in the models (Table 9.2). The models showed evidence of a slight effect of water temperature, DO, change in DO, flow and previous activity. The models suggested a peak activity when temperatures were around 24.3° C (23.9° C to 24.9° C 95% HDI) and 22.9° C (22.7° C to 23.1° C 95% HDI) for Murray cod and golden perch respectively. However, the influence of these variables was less than 10% of the unexplained variation (Table 9.2). In other words, any effect of these variables was relatively minor or weak compared to the overall variation in the dataset.

The depths of Murray Cod and Golden Perch ranged from 0.1-2.5 m (median 1.2 m) and 0.1-3.2 m (median 1.2 m), respectively (Figure 9.12, Figure 9.13). Both species tended to be closer to the surface at night (Murray Cod: median 1.1 m, Golden Perch: median 1.0 m) compared to the day (Murray Cod: median 1.3 m, Golden Perch: median 1.4 m) (Figure 9.12 and Figure 9.13).

Depth was not strongly influenced by any of the environmental variables examined for either species (Table 9.3). While the models showed evidence of a slight effect of water temperature, DO, flow and change in DO, the influence of most variables was less than about 5% and 12% of the unexplained variation for Murray Cod and Golden Perch, respectively (Table 9.3). In other words, any effect of these variables was relatively minor or weak compared to the overall variation in the dataset.

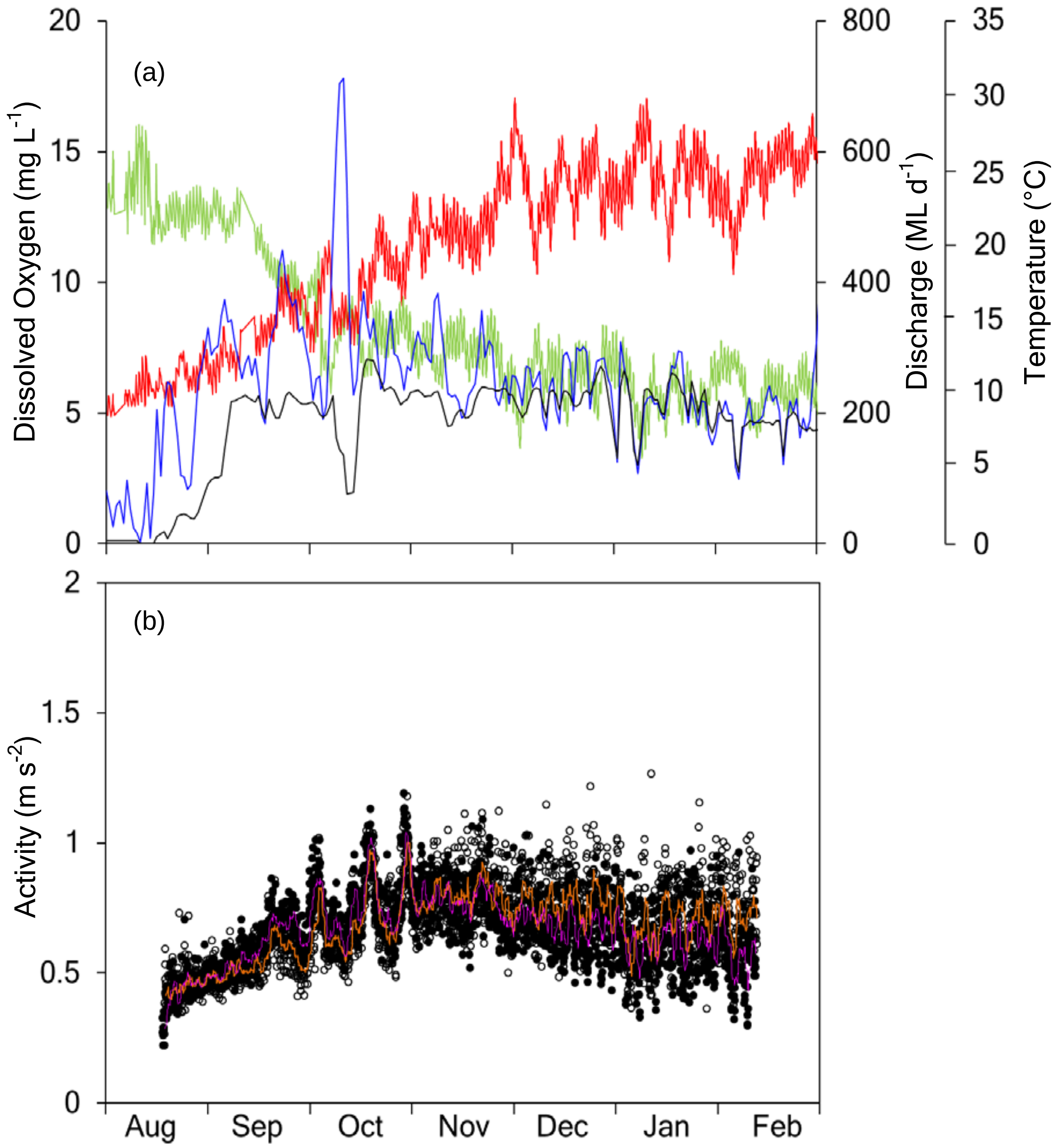


Figure 9.10: Hourly median activity patterns of Murray Cod in Broken Creek. (a) Blue line represents daily mean discharge, red line represents water temperature and green line represents dissolved oxygen in Broken Creek at Rices Weir. Black line represents total environmental water released. (b) Black circles represent night and white circles represent day. Pink and orange lines represent moving average during night and day, respectively.

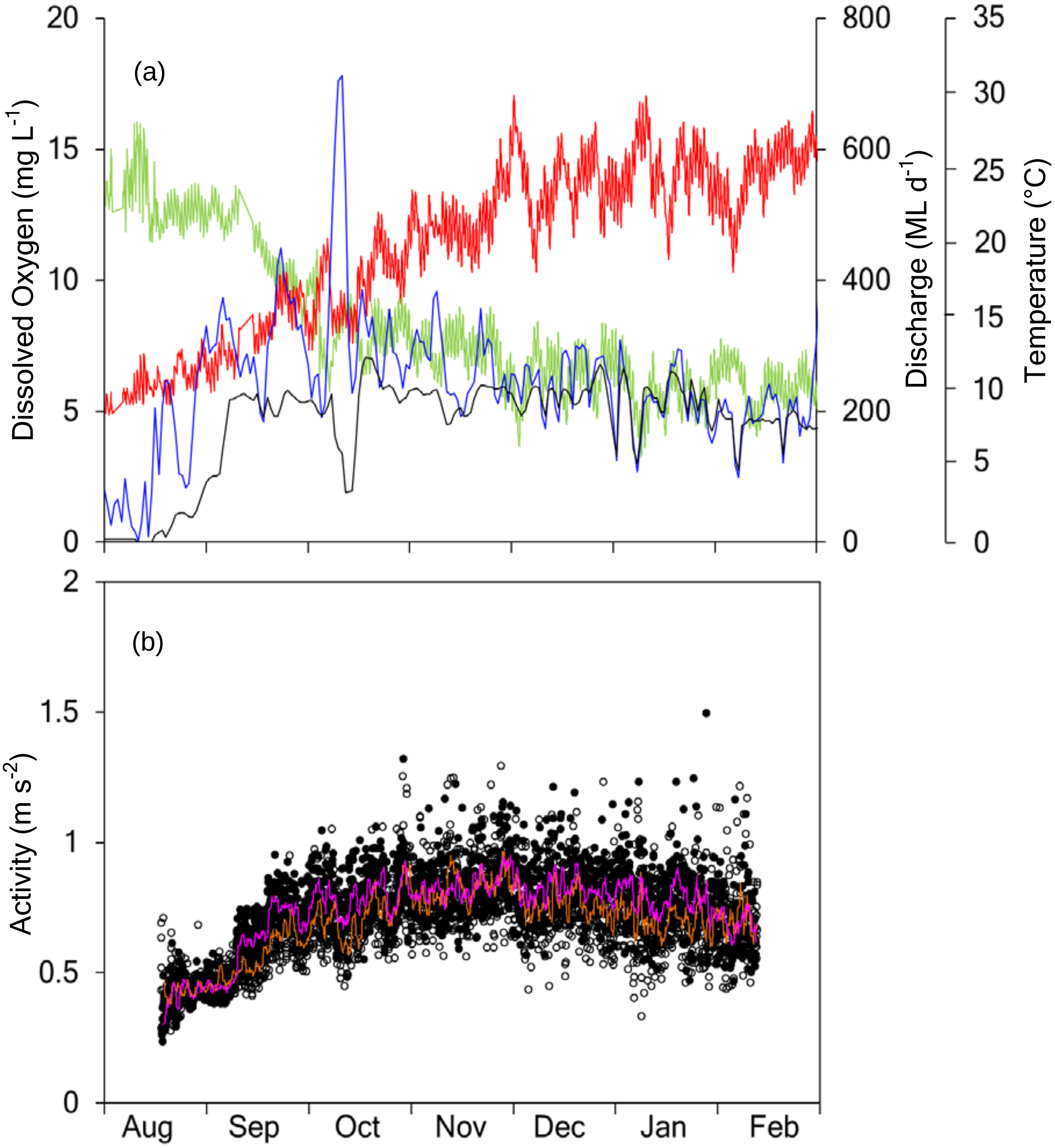


Figure 9.11: Hourly median activity patterns of Golden Perch in Broken Creek. Refer to Figure 9.10 for legend.

Table 9.2 Influence of environmental variables on activity of (a) Murray Cod and (b) Golden Perch in Broken Creek. Lower Bound – lower 95% high density interval. Upper Bound – upper 95% high density interval.

Parameter	Estimate	Lower Bound	Upper Bound	Estimate as a % of standard deviation
(a) Murray Cod				
Previous activity	0.115	0.110	0.119	35.4%
Water temperature	0.017	0.012	0.022	5.3%
Water temperature squared	-0.019	-0.023	-0.016	5.9%
Dissolved oxygen	0.028	0.023	0.033	8.7%
Change in DO	0.028	0.024	0.032	8.5%
Flow	0.025	0.020	0.029	7.7%
Change in Flow	-0.003	-0.007	0.001	1.0%
Mean activity, given average conditions	0.751	0.711	0.787	
Standard deviation (Overall)	0.324	0.321	0.327	
(b) Golden Perch				
Previous activity	0.070	0.066	0.074	24.9%
Water temperature	0.028	0.024	0.033	10.0%
Water temperature squared	-0.007	-0.010	-0.004	2.5%
Dissolved oxygen	0.028	0.024	0.033	9.9%
Change in DO	-0.011	-0.015	-0.007	3.9%
Flow	0.020	0.015	0.024	7.0%
Change in Flow	0.000	-0.003	0.004	0.2%
Mean activity, given average conditions	0.769	0.715	0.821	
Standard deviation (Overall)	0.282	0.280	0.285	

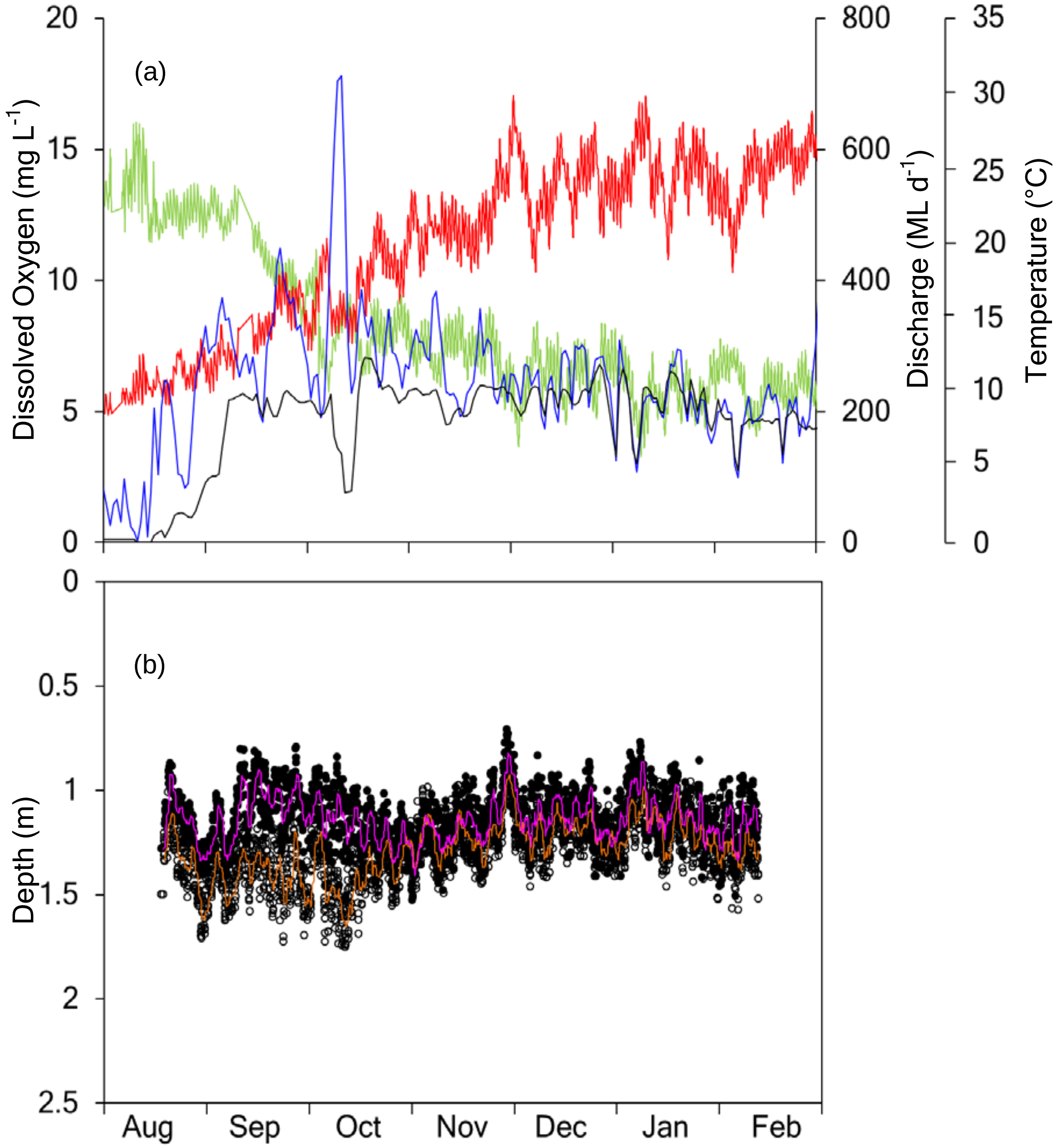


Figure 9.12: Hourly median depth patterns of Murray Cod in Broken Creek (Depth refers to distance below the water surface). Refer to Figure 9.10 for legend.

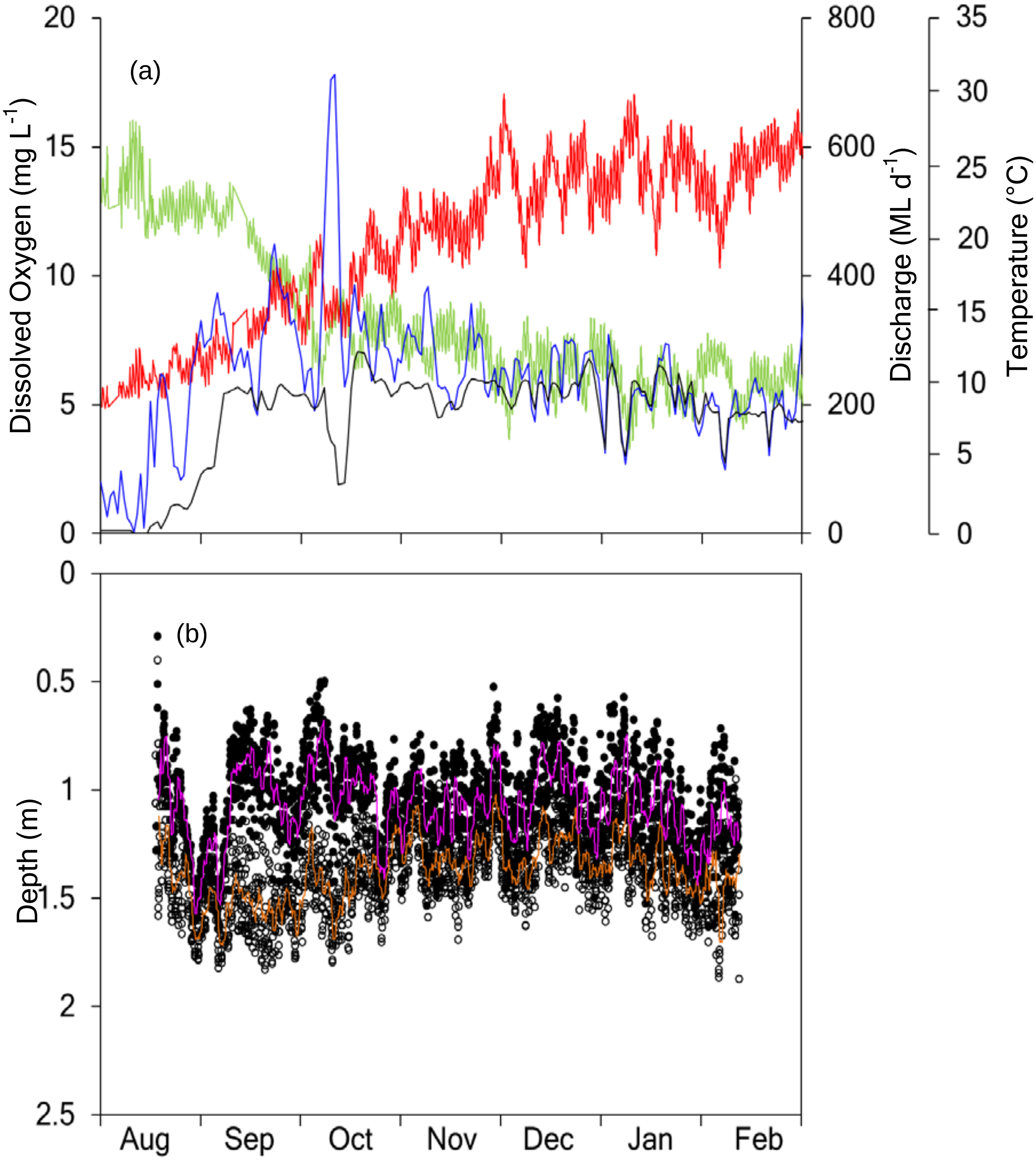


Figure 9.13: Hourly median depth patterns of Golden Perch in Broken Creek (Depth refers to distance below the water surface). Refer to Figure 9.10 for legend.

Table 9.3: Influence of environmental variables on depth (below water surface) of (a) Murray Cod and (b) Golden Perch in Broken Creek. Lower Bound – lower 95% high density interval. Upper Bound – upper 95% high density interval.

Parameter	Estimate	Lower Bound	Upper Bound	Estimate as a % of standard deviation
(a) Murray Cod				
Water temperature	-0.067	-0.070	-0.064	35.8%
Dissolved oxygen	-0.007	-0.010	-0.004	3.7%
Flow	-0.004	-0.006	-0.001	2.1%
Previous depth	0.288	0.285	0.292	154.0%
Change in DO	0.007	0.004	0.009	3.7%
Change in flow	0.000	-0.003	0.002	0.0%
Standard deviation (Overall)	0.187	0.185	0.188	
Standard deviation (Fish)	0.089	0.058	0.142	
(b) Golden Perch				
Water temperature	-0.061	-0.064	-0.057	31.8%
Dissolved oxygen	-0.023	-0.026	-0.020	12.0%
Flow	-0.011	-0.014	-0.008	5.7%
Previous depth	0.351	0.347	0.355	182.8%
Change in flow	0.018	0.015	0.021	9.4%
Change in flow	-0.003	-0.006	-0.001	1.6%
Standard deviation (Overall)	0.192	0.191	0.194	
Standard deviation (Fish)	0.075	0.050	0.116	

Longitudinal movements of Murray Cod and Golden Perch

Most Murray Cod remained close to their capture point and occupied small lengths of stream (median total linear range: 2.1 km) (e.g. Figure 9.14). However, five of the six (80%) Murray Cod ≥ 650 mm TL undertook occasional larger (> 5 km) upstream and downstream movements, particularly around September-October (e.g. Figure 9.15 and Figure 9.16). Only one (6%) smaller (< 650 mm TL) Murray Cod moved > 5 km. Movements of Murray Cod > 5 km typically coincided with flow rises (i.e. from 185 to 449 ML d^{-1} in September, 190 to 705 ML d^{-1} in October) (Figure 9.15, Figure 9.16 an Figure 9.17). One of the larger Murray Cod undertook an extensive (i.e. > 30 km) upstream shift away from its usual location including through Kennedys and Schiers Weirs in September, before returning to the area it previously occupied in November (Figure 9.16). These upstream and downstream movements coincided with flow rises (i.e. from 185 to 449 ML d^{-1} in September, 192 to 357 ML d^{-1} in November).

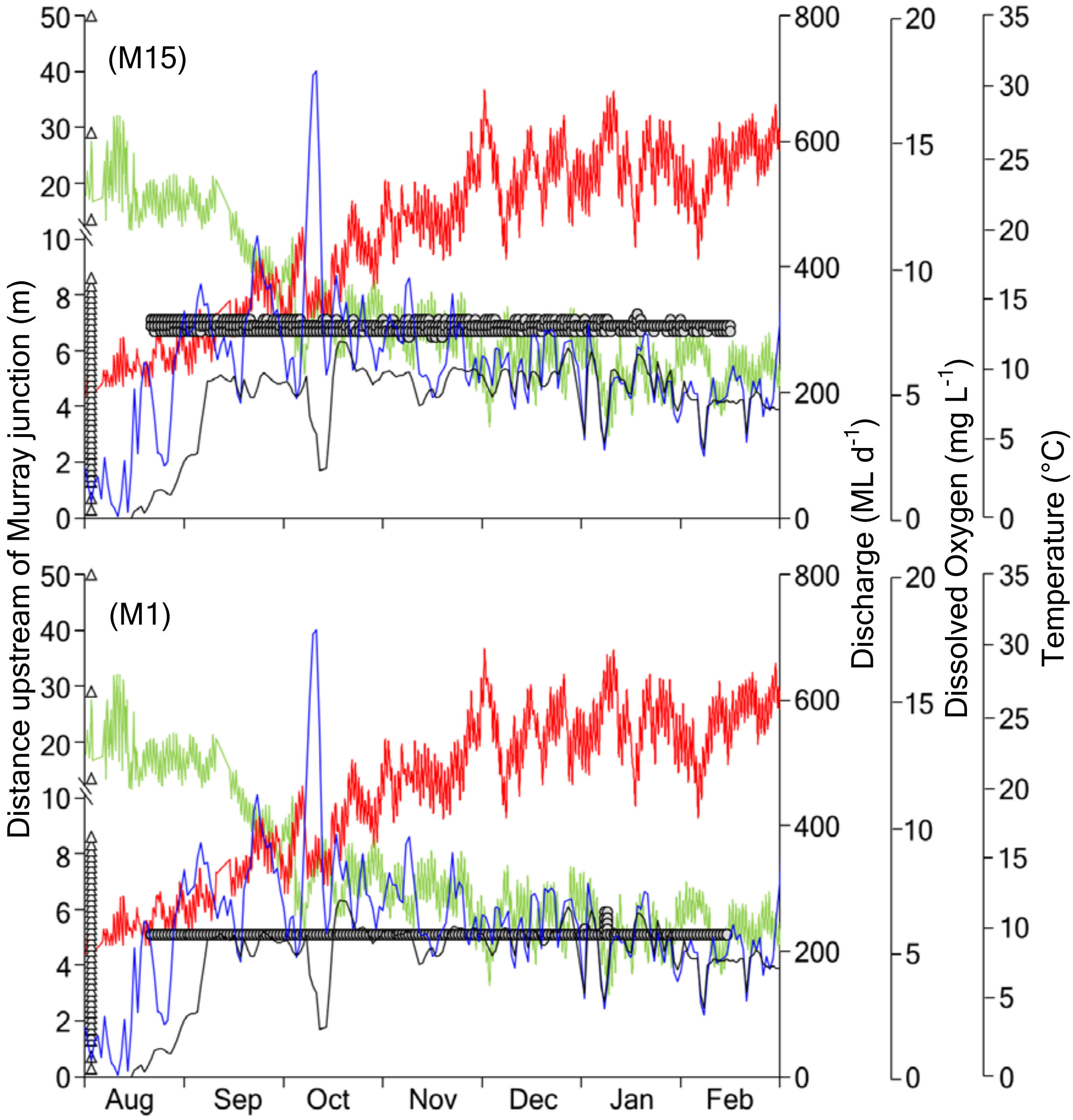


Figure 9.14: Examples of restricted movements by Murray Cod in Broken Creek. White triangles represent each of the listening stations. Grey circles show detections of fish on the listening stations. Blue line represents daily mean discharge, red line represents water temperature and green line represents dissolved oxygen in Broken Creek at Rices Weir. Black line represents total environmental water released. Letter/number refers to individual fish and correspond to the same fish as in Table 9.1.

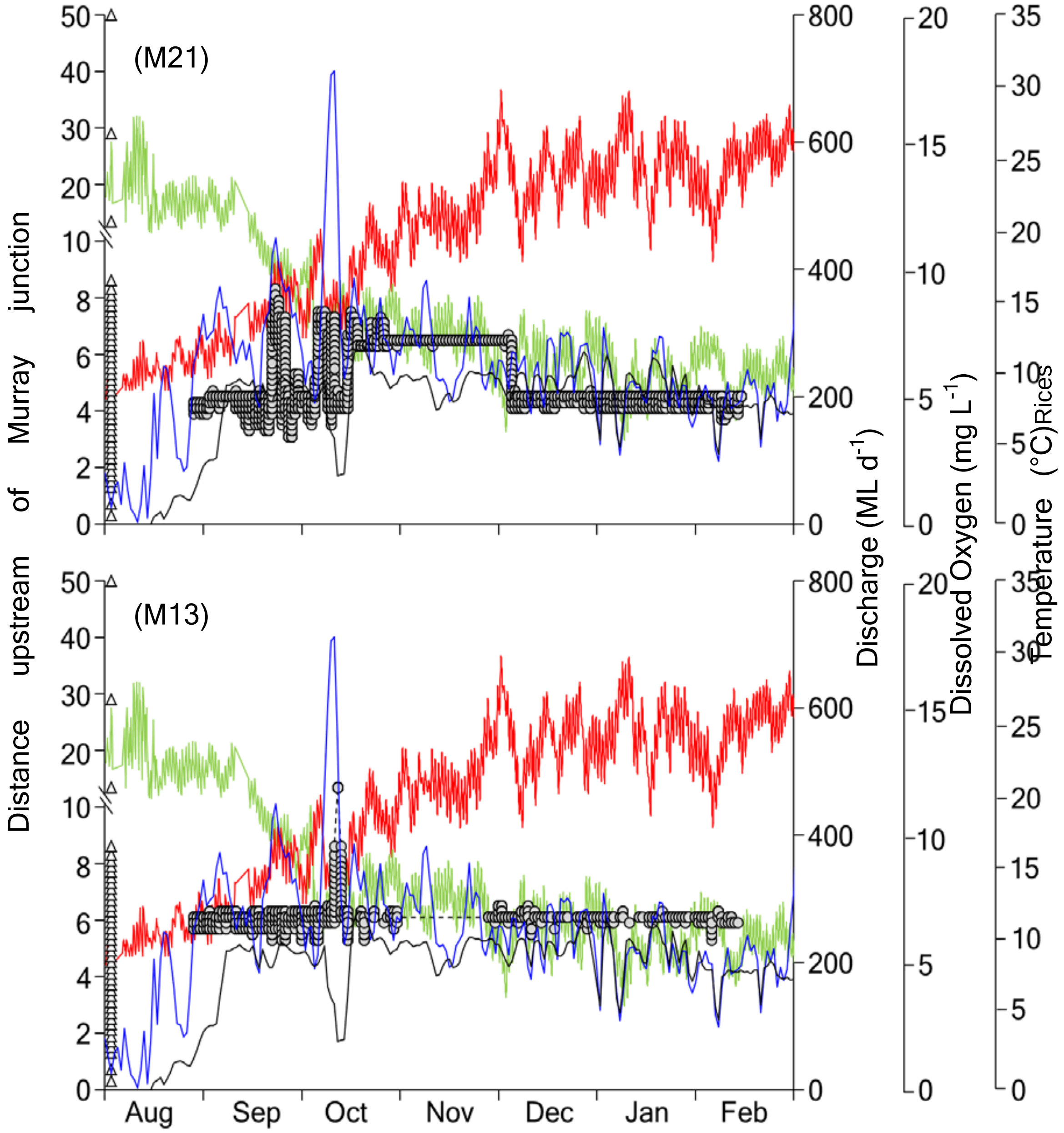


Figure 9.15: Examples of larger distance movements by Murray Cod in Broken Creek. Refer to Figure 9.14 for legend.

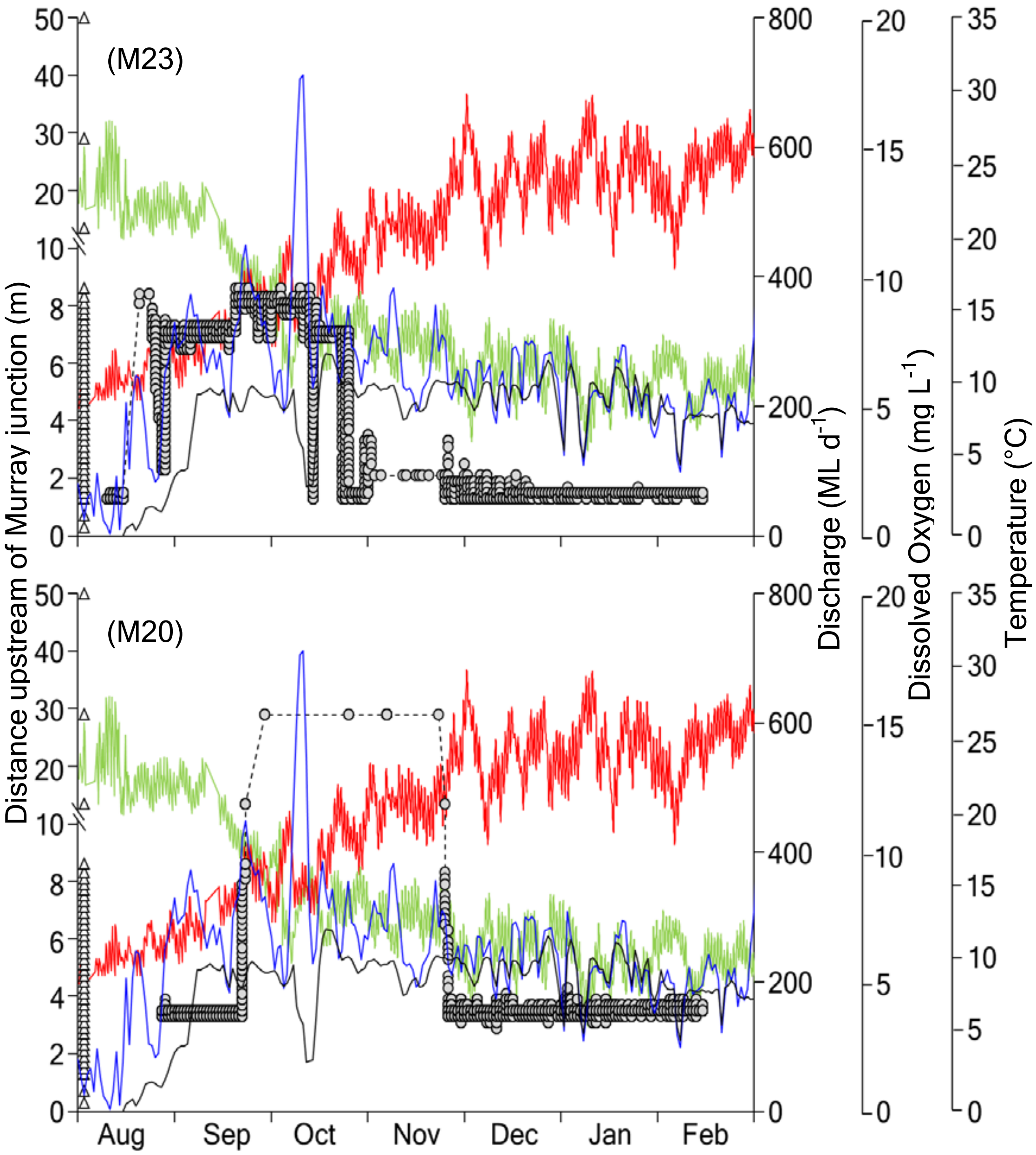


Figure 9.16: Examples of larger distance movement by Murray Cod in Broken Creek. Refer to Figure 9.14 for legend.

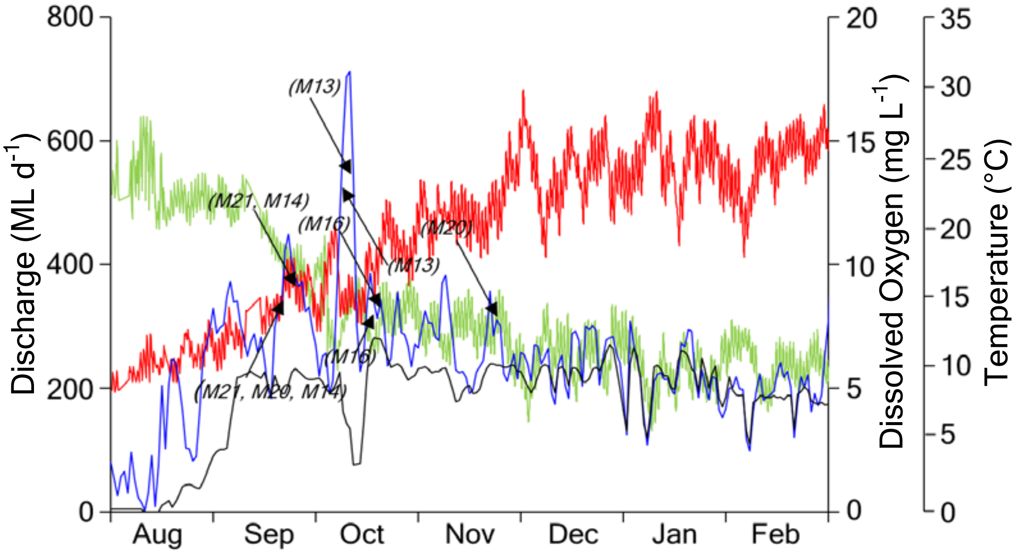


Figure 9.17: Approximate initiation of upstream (↗) or downstream (↘) movement (> 5 km) by Murray Cod in Broken Creek. Letter/number refers to individual fish and corresponds to the same fish as in Table 9.1. Blue line represents daily mean discharge, red line represents water temperature and green line represents dissolved oxygen in Broken Creek at Rices Weir. Black line represents total environmental water released.

Similar to the Murray cod, most Golden Perch remained close to their capture point and occupied small lengths of stream (median total linear range: 3.0 km) (e.g. Figure 9.18). Two Golden Perch undertook extensive (i.e. 30-50 km) upstream shifts including through Kennedys, Schiers and Hardings Weirs in November-December, where they remained throughout the study (Figure 9.19). These upstream movements occurred a few days after a flow peak, although water levels had dropped back to around baseflow levels (i.e. from 371 to 192 ML d^{-1} in November, 251 to 174 ML d^{-1} in December) (Figure 9.19 and Figure 9.21). Two Golden Perch also moved downstream including through Rices Weir fishway into the Murray River (Figure 9.20). These movements coincided with flow rises (i.e. from 83 to 289 ML d^{-1} in September, 108 to 223 ML d^{-1} in January) (Figure 9.20 and Figure 9.21). One of these Golden Perch was last detected moving through a fishway at Yarrawonga Weir in the Murray River in mid-November.

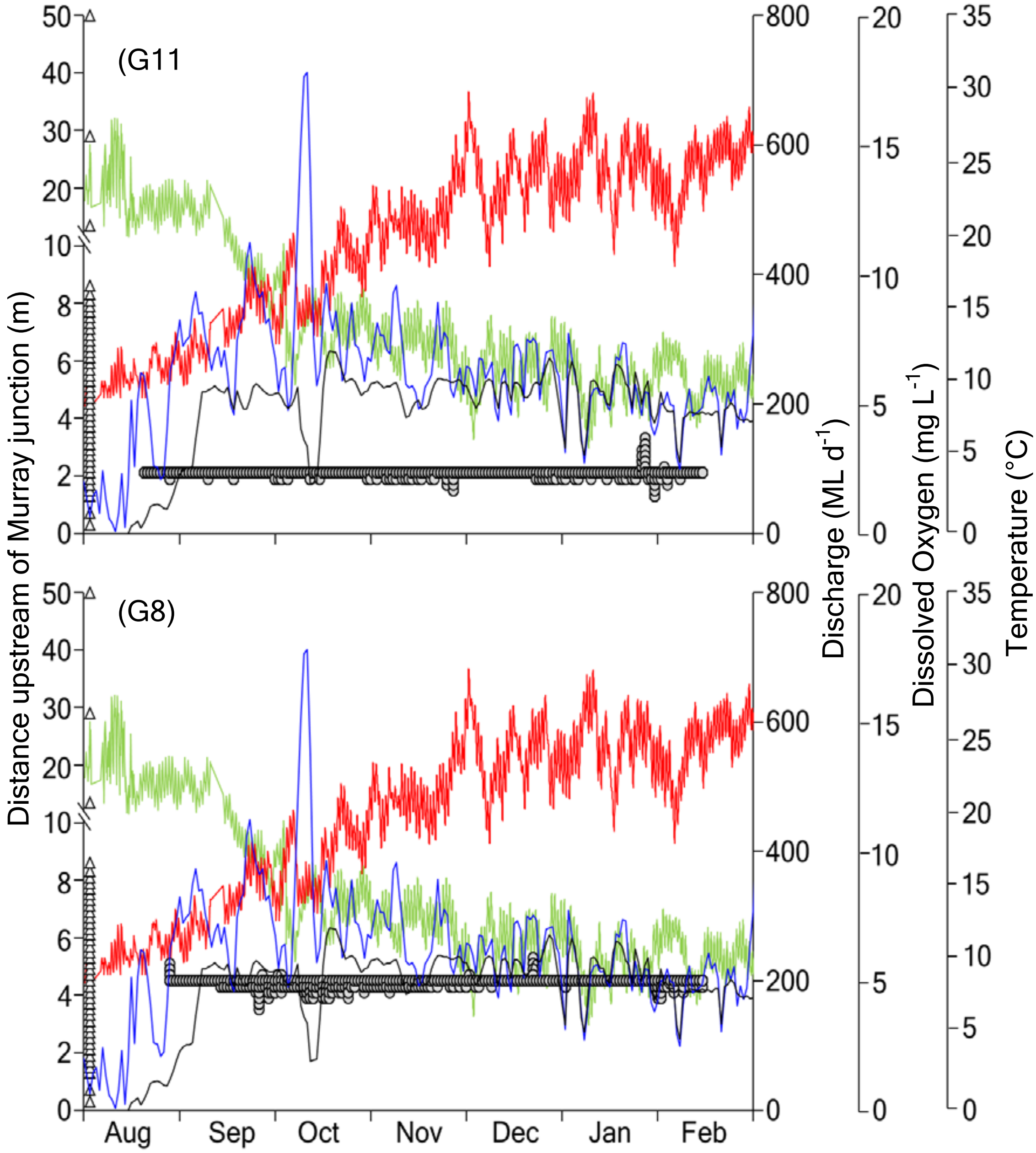


Figure 9.18: Examples of restricted movements by Golden Perch in Broken Creek. White triangles represent each of the listening stations. Grey circles show detections of fish on the listening stations. Blue line represents daily mean discharge, red line represents water temperature and green line represents dissolved oxygen in Broken Creek at Rices Weir. Black line represents total environmental water released. Letter/number refers to individual fish and correspond to the same fish as in Table 9.1.

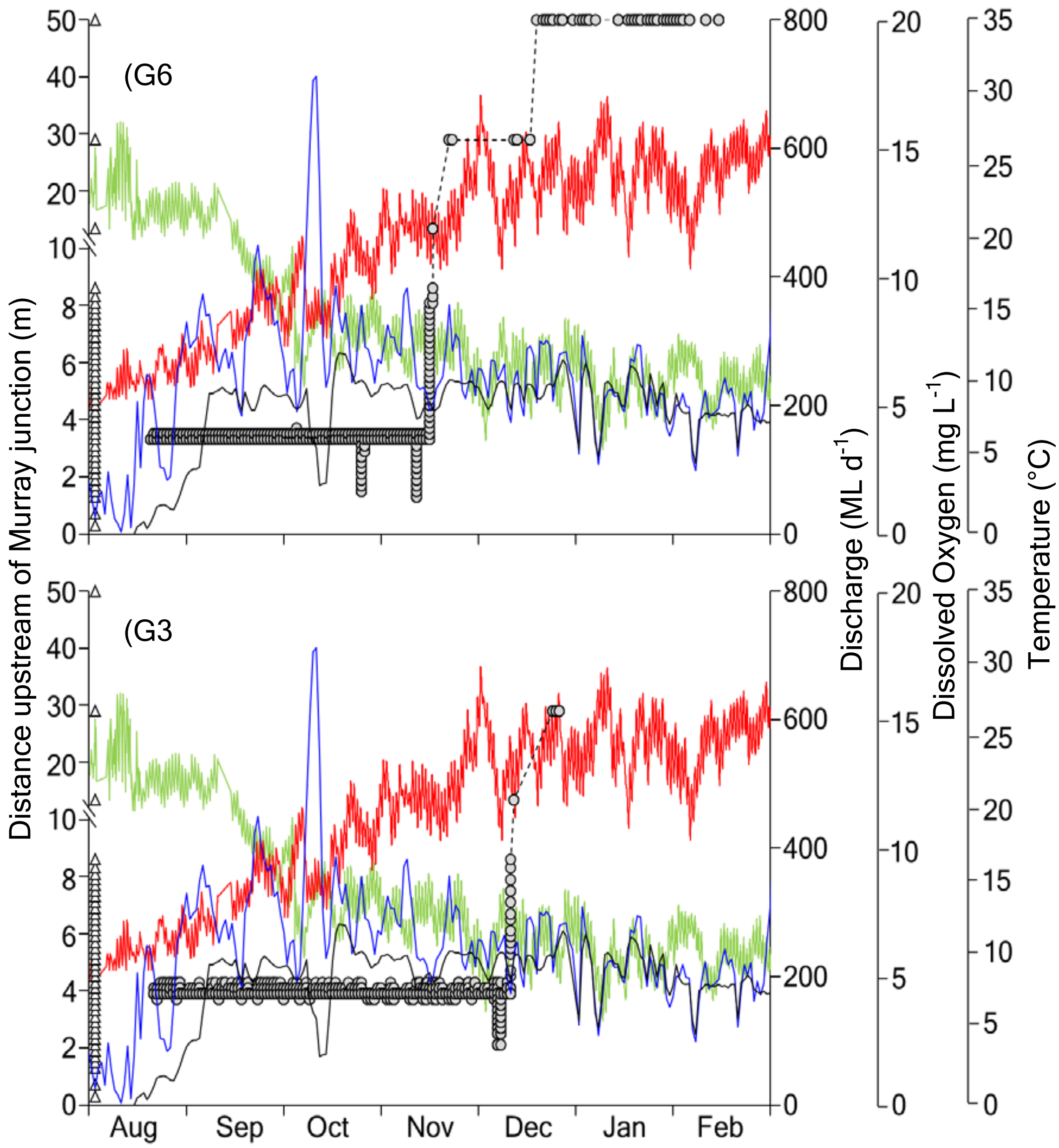


Figure 9.19: Examples of larger distance movements by Golden Perch in Broken Creek. Refer to Figure 9.18 for legend.

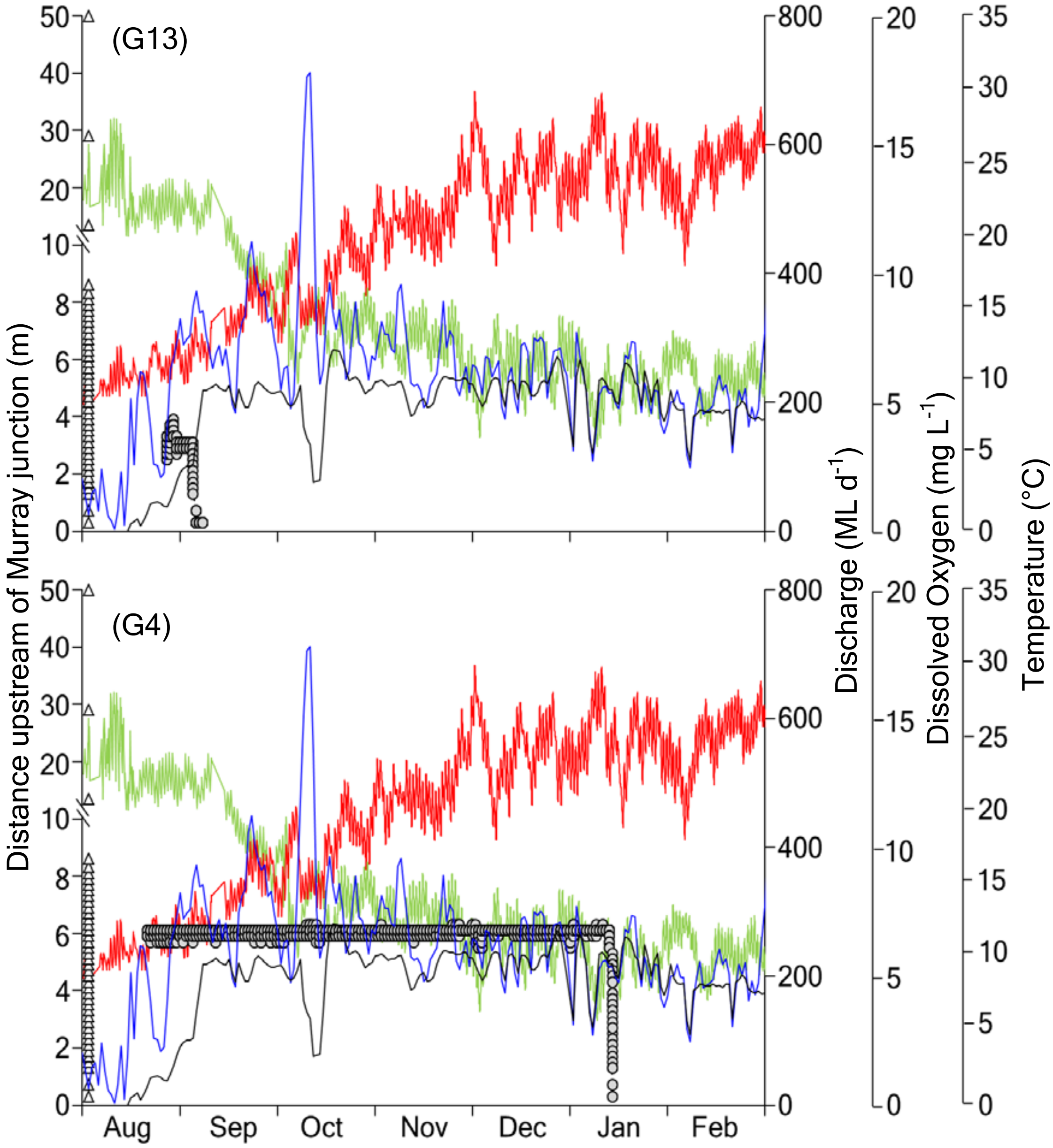


Figure 9.20: Movements by Golden Perch in Broken Creek into the Murray River. Refer to Figure 9.18 for legend.

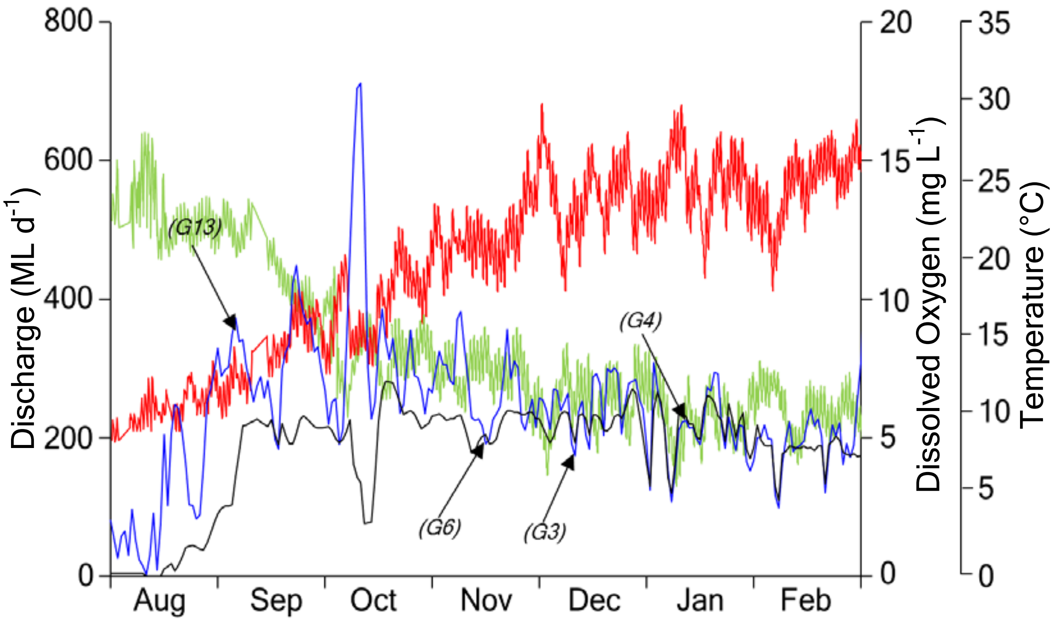


Figure 9.21: Approximate initiation of upstream (↗) or downstream (↘) movement (> 5 km) by Golden Perch in Broken Creek. Number/letter refers to individual fish and corresponds to the same fish as in Table 9.1. Blue line represents daily mean discharge, red line represents water temperature and green line represents dissolved oxygen in Broken Creek at Rices Weir. Black line represents total environmental water released.

9.4 Discussion

Activity and depth patterns of Murray Cod and Golden Perch

A major fish behavioral response to reduced dissolved oxygen can include vertical habitat changes or changes in activity (Kramer, 1987). For example, when vertical DO stratification occurs, fish may move to the surface of the water column to access the generally higher oxygen levels nearer the surface (Kramer, 1987, McNeil and Closs, 2007). During low DO events, fish may also either decrease or increase their swimming speed; demersal or sedentary species tend to reduce their activity, whilst active, schooling species tend to increase their activity (Herbert et al., 2011, Poulsen et al., 2011). Under this scenario, we might have expected Murray Cod, considered largely a sedentary species, to reduce activity, and Golden Perch, considered a more active species, to increase activity, under low DO conditions.

Murray Cod and Golden Perch activity was not strongly affected by DO in Broken Creek. This result may reflect the finding that DO concentrations in Broken Creek generally remained above 4 mg L^{-1}. Previous research indicates hypoxic stress in fish is typically evident once DO concentrations fall below 4 mg L^{-1} (Gehrke, 1988, McNeil and Closs, 2007). Although DO concentrations occasionally decreased to about 2.5-3 mg L^{-1}, these events were short-lived (i.e. < 1-2 d duration). Time spent in low DO waters (in addition to the level of DO) is thought to be a major factor affecting behaviour and survival of fish exposed to low DO (Herbert et al., 2011).

Neither Murray Cod nor Golden Perch displayed shifts to the water surface in response to DO. This result possibly reflects the finding that DO concentrations in Broken Creek at bottom and surface were virtually identical. Although there were occasional periods when bottom DO was ~ 2-3 mg L^{-1} lower than surface, these events were short-lived (i.e. < 1 d duration). As previously mentioned, resistance of fish to hypoxia is likely to also be a function of the length of hypoxia (Herbert et al., 2011).

Due to an absence of persistent hypoxic events in Broken Creek over spring-summer 2012/13, it remains unclear whether Murray Cod and Golden Perch exhibit stress responses during such periods. Further assessment of whether Murray Cod and Golden Perch move higher up in the water column to avoid DO depleted bottom waters would be valuable to inform the provision of environmental flows to improve water quality for the Broken Creek. For example, depending on the behavioural responses of the fish, if DO stratification occurs in a pool, it may be desirable to release a large volume of water to completely flush out the pool, or alternatively a smaller volume that flushes only the surface water might be sufficient.

Whilst Murray Cod and Golden Perch did not move to the surface in response to DO, vertical shifts were observed over the diel (day-night) period. In particular, both species tended to move closer to the surface at night compared to during the day. Similarly, in the Broken River, Golden Perch used deeper habitats during the day compared to night (Crook et al., 2001). Although Murray Cod have been shown to use positions low in the water column during the day in the Ovens River (Koehn, 2009), the current study adds to knowledge of vertical habitat use of the species by demonstrating a shift towards the surface at night. Diel habitat shifts are typically associated with feeding opportunities and predator avoidance (Clapp et al., 1990, Roussel and Bardonnet, 1997) and the results serve to highlight the importance of providing adequate flows to maintain a diversity of habitats for Murray Cod and Golden Perch in Broken Creek.

While activity of Murray Cod and Golden Perch was not strongly influenced by any of the environmental variables examined, Murray Cod activity did exhibit several pronounced spikes in October, coinciding with sudden rises in temperature (e.g. from 16.2 to 23.2° C). The timing of these activity spikes coincides with the spawning season of Murray Cod (Koehn and Harrington 2006). Spawning of Murray Cod occurs at the same approximate time each year (September-October) when water temperatures are about 16-20° C (Koehn and Harrington 2006). On the basis of these results, it is likely that the increased activity by some of the Murray Cod in October may have been be related to their reproductive behaviour.

Longitudinal movements of Murray Cod and Golden Perch

Another fish behavioral response to reduced availability of dissolved oxygen can include moving away from areas affected by low DO concentrations (Kramer, 1987, Knights et al., 1995). For example, in November 2010 increased abundances of Golden Perch and Silver Perch (*Bidyanus bidyanus*) were collected in the lower Goulburn River near the Murray River junction following a “blackwater” (i.e. de-oxygenation) event in the Murray River, and it was suggested that fish may have moved into the Goulburn River as a refuge (King et al., 2012).

We did not observe synchronised movement of Murray Cod or Golden Perch away from the study area in Broken Creek, with the vast majority of fish occupying restricted home ranges. Most (5 of 6) of the larger Murray Cod undertook occasional larger (> 5 km) upstream and downstream movements, mostly around September-October, with one fish travelling ~30 km upstream and downstream, including through several fishways. Several Golden Perch also shifted away from their usual locations, with two fish moving 30-50 km upstream, including through several fishways, and two fish moving downstream, including through Rices Weir fishway, and into the Murray River. Although there were short periods (i.e. < 1-2 d duration) when DO decreased to about 2.5-3 mg L^{-1}, a level potentially harmful to fish (Gehrke, 1988, McNeil and Closs, 2007), these periods of reduced DO did not coincide with the larger movements.

For Murray Cod, the timing of the larger movements coincides with the spawning season (Koehn and Harrington, 2006). These findings are consistent with the results of previous studies that found Murray Cod are sedentary for extended periods (Koehn et al., 2009, Reynolds, 1983), but fish > 650 mm TL, which likely represent mature adults, may undertake larger (> 5 km) upstream or downstream movements around the spawning period, possibly to find suitable spawning sites or mates (Koehn et al., 2009). In contrast to the study by Koehn *et al.* (2009), however, which found 85% of fish > 650 mm TL in the Ovens and Murray Rivers moved further than 10 km, with many moving > 30 km, only 1 fish > 650 mm TL moved more than 10 km in the present study. This may suggest that Murray Cod in the lower Broken Creek are able to utilise ‘locally’ available spawning habitat, negating the need for large (> 10 km) scale spawning movements. In contrast, many of the Murray Cod examined by Koehn et al. (2009) were tagged in an artificial lake (Lake Mulwala) where they may have had to migrate upstream to access suitable riverine spawning habitat. Use of techniques such as larval drift net sampling would be valuable to determine the location and spatial extent of Murray Cod spawning habitat in Broken Creek.

Larger (i.e. > 5 km) distance movements by Murray Cod typically coincided with changes in flow (e.g. from 185 to 449 ML d^{-1}, 190 to 705 ML d^{-1}). These changes in flow were driven by rainfall runoff from the catchment, rather than delivery of environmental water to Broken Creek, which was delivered as a relatively constant discharge (e.g. 200 or 250 ML d^{-1}). Nonetheless, the results have important implications for the provision of environmental flows designed to facilitate movement of Murray Cod throughout Broken Creek. In particular, the results suggest that delivery of environmental water which includes variations in flow, or 'freshes', may be more desirable than providing a constant discharge, particularly in the absence of rainfall runoff events. Given the limited number of Murray Cod that undertook large scale movements in the current study, further assessment of movement patterns over a more extensive period would be valuable to better understand the role of flows in the movement of Murray Cod. For example, movements coincident with river flow were not detected in a study of Murray Cod movement in the Ovens and Murray Rivers (Koehn, 2009).

Whilst the allocation of environmental water to Broken Creek did not include significant flow variations, the magnitude of flow was nonetheless increased. Changes in flow are often associated with movement of fish, but movement may also be dependent on a certain magnitude of flow, possibly in combination with variations in flow (Erkinaro et al., 1999, Ovidio et al., 1998, Baran, 2006). In the present study, larger movements of Murray Cod coincided with changes in flow, but only when flow was above about 200 ML d^{-1}; whether these movements would have occurred without the increase in flow magnitude provided by the allocation of environmental water is unclear. For example, there were some variations in flow at lower magnitudes (e.g. from 108 to 220 ML d^{-1} in January, 99 to 223 ML d^{-1} in February), but these occurred outside of the spawning season, and hence larger movements might not have been be expected to occur.

For Golden Perch, the timing of the larger movements was non-synchronous among fish (September to January), which suggests that such movements were not related to a specific life history event (e.g. reproduction). Shifts away from established locations have previously been reported for Golden Perch and it has been suggested that such movements may relate to exploration and assessment of new habitat (Crook, 2004a, Crook, 2004b).

The shifts by Golden Perch tended to coincide with changes in flow. In particular, downstream movements coincided with flow rises whilst upstream movements occurred shortly (~ 1 week) after flow rises. These changes in flow were driven by rainfall runoff from the catchment (e.g. from 83 to 289 ML d^{-1}) or resulted from the use of water for irrigation (e.g. from 251 to 174 ML d^{-1}). These findings support previous research that suggests Golden Perch movement is associated with changes in flow (O'Connor et al., 2005, Reynolds, 1983) and serves to demonstrate the role of flows in facilitating Golden Perch movement along Broken Creek, and into the Murray River. Similar to Murray Cod, only a small proportion of tagged Golden Perch undertook larger movements and further assessment of movement patterns over a more extensive period would be valuable to better understand the role of flows in the movement of this species. For example, previous studies have suggested that Golden Perch may undertake long (e.g. 10s-100s of km) distance spawning migrations during increased flow in spring (Reynolds, 1983, O'Connor et al., 2005), but there was little evidence of such migratory behaviour in the current study.

Conclusions

Extended periods of low dissolved oxygen did not occur during the study period, limiting inference on the responses of fish to extreme low DO events in Broken Creek. Nonetheless the study has provided valuable baseline information on fish movement responses to small, short-term changes in DO and flows that provides a basis on which to assess the potential threats or effects of any future events. In particular, the study has shown that short term oxygen depletion to about 2.5-3 mg L^{-1} in Broken Creek did not appear to affect behaviour of fish as measured by their levels of activity, position in the water column, or longitudinal movements. If fish are exposed to more extensive periods of DO depletion, however, their responses are likely to differ. Understanding the thresholds at which stress responses are elicited remains an important area for future research given hypoxic events have become increasingly frequent in waterways in the Murray-Darling Basin such as Broken Creek.

Longitudinal movements of Murray Cod and Golden Perch were associated with hydrological events. In particular, changes in flow (i.e. freshes) appear to be important for promoting the movement of these species throughout Broken Creek, and also into the Murray River. These results have important implications for the provision of environmental flows designed to facilitate movement of Murray Cod and Golden Perch. Delivery of environmental water to Broken Creek that includes variations in flow may be more desirable for promoting fish movement than providing a constant discharge.

The V9 transmitters (n = 28) implanted into fish will likely expire around mid-2013. The V13 transmitters (n = 10) should continue to transmit till around mid-2014 and thus could allow for further assessment of the behaviours of fish over another spring-summer (2013/14). Notwithstanding, only 10 tagged fish will have active transmitters during spring-summer 2013/14 and therefore extending the acoustic tracking study by tagging a further 15-20 fish would be valuable to improve understanding of fish movement responses to dissolved oxygen and flows.

10 FISH MOVEMENT THROUGH FISHWAYS

10.1 Evaluation question and expected outcomes

This section addresses the evaluation question: Does fish movement through fishways increase with increasing river discharge and does it increase from September to January? The expected outcome was that increasing flow through environmental releases would promote increased movement of fish through fishways.

10.2 Methods

Background

Longitudinal connectivity is important for maintaining native fish populations (Mallen-Cooper, 1996), and instream barriers are thought to adversely affect the movement behavior of fish (Jones and Stuart, 2008). With this in mind, vertical slot fishways were constructed on each of the low-level weirs along Broken Creek between Barmah and Numurkah during the late 1990s and early 2000s. Further, Environmental Water Allocations (EWA) are now being used to help mitigate the adverse effects of weir/dam construction and flow alteration on instream biota along Broken Creek.

Prior research has shown that rising discharge may be particularly important in September-October to stimulate primary production and development of larval food sources within a river system (Junk et al. 1989), while this rising discharge may also stimulate pre-spawning migrations of Murray cod and golden perch (Koehn et al. 2009, O'Connor et al. 2005). Additionally, creek rises in mid-November may also facilitate Murray cod larval drift (Zeb Tonkin pers. Comm.), an important life-history component, and potentially golden perch spawning (Koster et al. 2012). Additional rises in creek discharge during December and January should aid dispersal of juvenile fish, and spawning of other large bodied native species such as silver perch.

This investigation monitors the movements of fish through the Broken Creek fishways (fishway trapping) throughout the spring-summer period 2012/13. In order to guide the project design and analysis, a conceptual model was developed using previous fish movement information from Broken Creek (O'Connor and Koster, 2005) and other regulated systems (Figure 10.1). The conceptual model proposes that elevated flows during the spring period will result in the highest number of fish moving through the Broken Creek fishways. Two hypotheses were developed from this: 1) The movement of fish through the Broken Creek fishways will increase with river discharge and, 2) The movement of fish through the

Broken Creek fishways will increase from September to January. These hypotheses were tested using fishway trapping data and fish movement data.

Study Site

Broken Creek flows for approximately 200 km in a westerly direction from Boosey Creek to the Murray River at Barmah (Figure 10.2). Ten low level weirs control flows along lower Broken Creek between Numurkah and Barmah, these were constructed to regulate flows for agricultural purposes. Mean daily discharge in the creek is approximately 230 ML/d (range zero to 2139 ML/d). Each of the ten low-level weirs has a vertical-slot fishway to pass fish.

Broken Creek is surrounded by agricultural land, however the stream bank is lined with river red gum (*Eucalyptus camaldulensis*), and Grey box (*Eucalyptus microcarpa*). Instream habitat consists of interspersed emergent cumbunghi (*Typha* spp.), spike rush (*Eleocharis* spp.), common reed (*Phragmites australis*), whilst large and small woody debris are frequently encountered.

A total of 13 fish species, eight native and five alien, have been recorded in the Broken Creek system (ARI, 2006, McKinnon and Shepheard, 1995, O'Connor and Amtstaetter, 2008, O'Connor and O'Mahony, 2008). Murray cod (*Maccullochella peellii*), golden perch (*Macquaria ambigua*), common carp (*Cyprinus carpio*) and goldfish (*Carassius auratus*) dominate the instream biomass, while Australian smelt (*Retropinna semoni*), Murray-Darling rainbowfish (*Melanotaenia fluviatilis*), carp gudgeon (*Hypseleotris* spp.), flathead gudgeon (*Philypnodon grandiceps*), unspecked hardyhead (*Craterocephalus stercusmuscarum fulvus*) and silver perch (*Bidyanus bidyanus*) are also encountered. Golden perch and Murray cod are also stocked into the creek (ARI 2006, Fisheries Victoria Management Report Series No. 82, June 2011).

Fishway design

A total of 10 vertical-slot fishways are constructed at weirs along lower Broken Creek between Numurkah and Barmah (Figure 10.2). The fishways are designed with a slot width of 30 mm, cell width of approximately 2 m and a baffle height of 1.2 m. Each fishway is generally operated from August to May, for the irrigation season, with all exit gates being remotely operated by Goulburn-Murray Water.

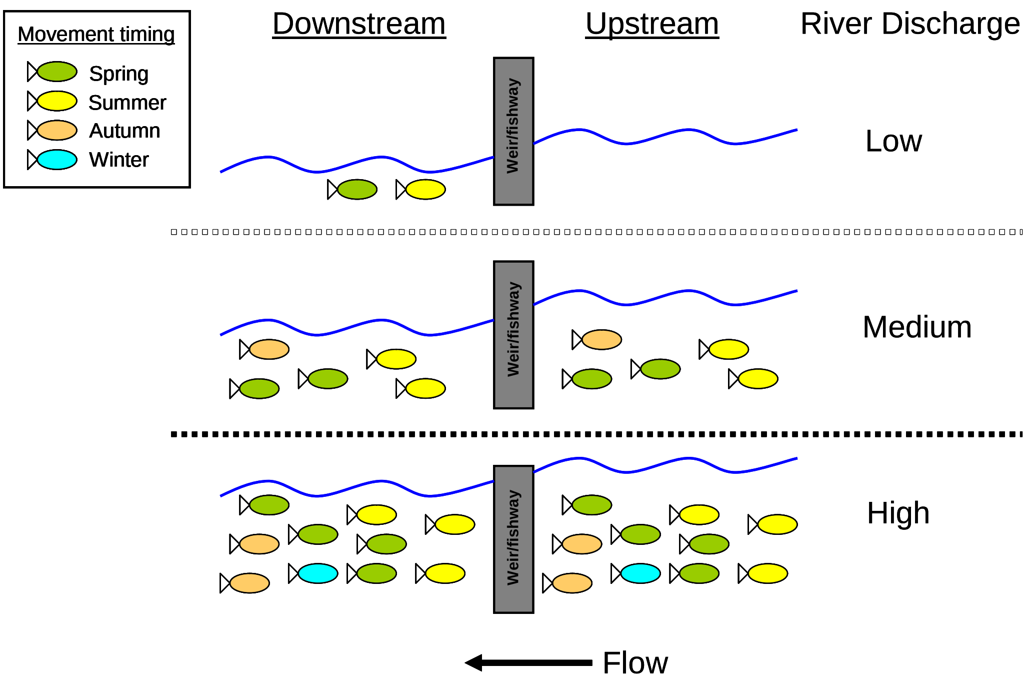


Figure 10.1: Conceptual model of fish movement through the Broken Creek fishways- more fish move through the Broken Creek fishways during high flows in the spring-summer period.

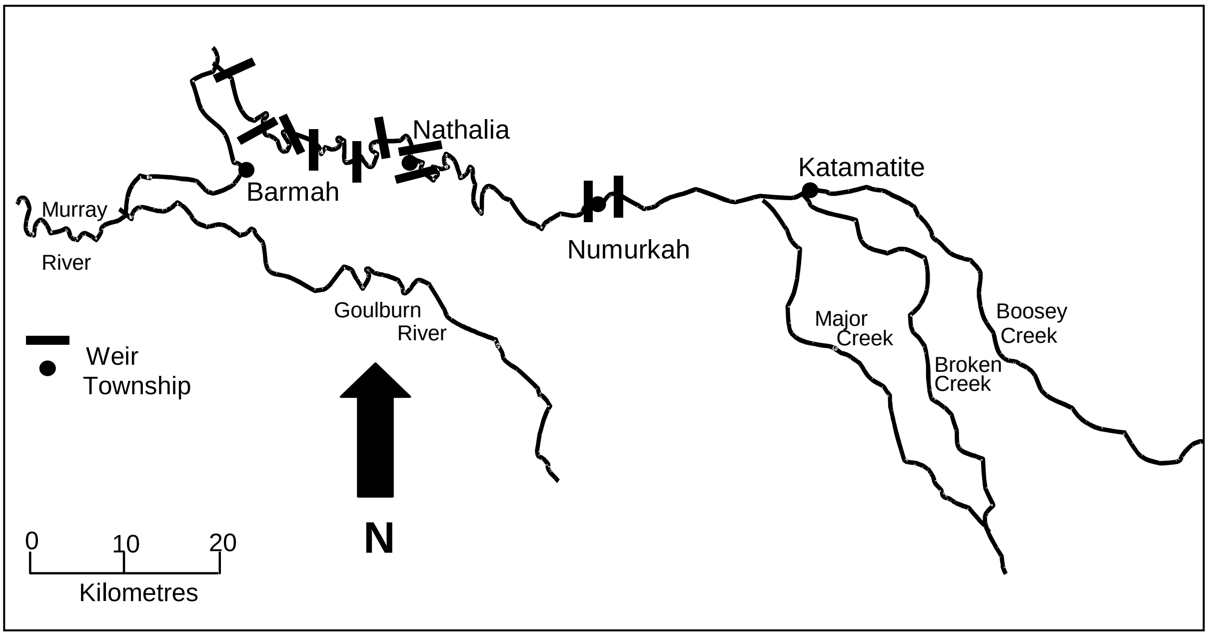


Figure 10.2: Map of study are showing townships and weirs marked as black bars (modified from (O'Connor and Amtstaetter, 2008)).

Figure 10.3: Schiers Weir with fishway and PIT tag system.

Trap and Sampling design

A power analysis was conducted on fish movement data previously collected from the Broken Creek fishways (O'Connor and Koster, 2005) in order to determine the sampling frequency required to detect a significant change in movement through the fishways. The analysis revealed a 74% chance of detecting such a change when eight fishways were sampled per trip, with a total of six trips; note, additional trips were preferred, but these were constrained by budgetary requirements. Thus, eight fishway traps were built to sample the eight fishways downstream of Nathalia.

Fishway traps were constructed of one inch square DuraGal® steel, and aluminium and galvanised mesh. Traps were 1.6 m long by 1.2 m high by 0.63 m wide (Figure 10.4). Aluminium perforated mesh (4 mm diam) was used on the floor to prevent damage to the fish. Each trap was covered with 50 mm galvanised mesh, to minimise the amount of debris collected on the trap; debris build-up on the trap alters the headloss through the fishway slot, and so limits smaller size-classes of fish from moving into the trap.

Fishway trapping was conducted monthly between September 2012 and March 2013 (Figure 10.4). Four traps were set on a Monday afternoon, and the remaining four were set on Tuesday morning; this sample design allowed enough time to drive between all trap sites. Each trap was set for 24 hours. All fish collected were counted and identified, with fish >200 mm in total length tagged with an external dart tag (with a unique identification number). Dart tags were used to identify fish upon recapture, and to direct anglers to record the capture location, fish length and whether it was removed from the waterway (i.e. angler instructions and phone number are on the tag). Passive Integrated Transponders (PIT) were also implanted into fish >200 mm, PIT tags allow fish to be recorded as they move through the network of PIT readers systems installed along Broken Creek. Fish implanted with PIT tags from previous research programs on Broken Creek (ARI, 2006, McKinnon and Shepheard, 1995, O'Connor and Amtstaetter, 2008, O'Connor and O'Mahony, 2008), were also included in the study. PIT tag movement data, creek discharge and water temperature were averaged by month (July 2009 to February 2012) for the PIT tag movement analysis – descriptive statistics were used.

Figure 10.4: a) Fishway trap placed into a fishway – water direction is from top of photo to bottom b) two golden perch and a common carp caught in a fishway trap.

Statistical Analysis

Bayesian analysis (Kery, 2010) was used to analyze the trapping/movement data. Three Bayesian models were considered for the most common species (Murray cod, common carp, golden perch), with all models to include flow, temperature and trip (i.e. sample date) as fixed variables. Site (or weir) was either excluded or included as a fixed or random effect for each model. The effect of the different weirs would then be judged using information criteria to select the model with the most evidence, however temperature and trip were deemed collinear [Generalised Variance Inflation Factors were greater than 4 (Fox and Monette, 1992)], therefore, models were generated using either temperature or trip, but not both. Consequently, six models were considered. The structure of the negative binomial model using flow and trip number as fixed effects and weir as a random effect is given below,

$$Catch_{i,j} \sim NB(p_{i,j}, size)$$

$$p_{i,j} = \frac{size}{size + \mu_{i,j}}$$

$$\ln(\mu_{i,j}) = \alpha + \beta \times Flow_{i,j} + \gamma \times Trip_i + \varepsilon_j$$

$$\varepsilon_j \sim N(0, \sigma^2_{Weir})$$

where *Catch* is the number of fish of a given species caught on trip *i* at weir *j* (Table 10.1). The model with a fixed effect for the weir would replace the random component with dummy variables for the weirs, while the model without weir effects removes the term. Models for temperature and binomial and Poisson distributions take an analogous form. Negative binomial models assumed a constant size parameter across all the data. Flow and temperature data was normalised to assist with computation and interpretation of results.

Each model was compared using the information criteria to determine which model had the most evidence using the most appropriate distribution of catch. Data from a previous study (O'Connor and Koster, 2005) was also used to inform the prior distribution. The models were constructed in JAGS (Plummer, 2003) via R (R Development Core Team, 2013) using the package R2jags (Su and Yajima, 2012). Model chains were run until the chains converged. Convergence was defined as having all Gelman and Rubin’s convergence diagnostic potential scale reduction factors being less than 1.05 (Gelman et al., 2004). If the 95% high density interval (HDI) did not include zero, the parameter was considered to influence the rate at which fish use the fishway. Assessing the model fit was done using posterior predictive Bayesian p-values and Pearson residuals. A posterior predictive Bayesian *p*- value close to 0 or 1 is evidence that the model does not fit the data very well (Gelman et al. 2004). Similarly Pearson residuals much greater than 4 indicate a poor fit.

Table 10.1: Bayesian Model definitions

Parameter	Definition
$Catch_{i,j}$	Number of fish caught on trip *i* at weir *j*.
$p_{i,j}$	Shape parameter for the negative binomial distribution on trip *i* at weir *j*.
$size$	Scale parameter for the negative binomial distribution.
$\mu_{i,j}$	Expected number of fish caught on trip *i* at weir *j*.
$Flow_{i,j}$	Water discharge on trip *i* at weir *j*. (Standardised)
$Trip_i$	Number of times the weir has been visited previously in this study.
$Temperature_i$	Temperature on trip *i*. (Standardised)
ε_j	Random effect term for weir *j*.
σ^2_{Weir}	Between weir variance.

10.3 Results

Water Discharge

Water discharge was obtained for weirs along lower Broken Creek between Rices and Nathalia town weir, however only three sites have been presented for visual comparison (Figure 10.5). River discharge generally ranged from approximately 150-300 ML/d at each weir throughout the study period, however substantially higher discharges were recorded in late September, early October, late February/early March. Of the three sites presented, discharge at Rices Weirs (last weir on the creek) was generally greater when compared with upstream sites and as such was used in the Bayesian analysis; note Nathalia Weir is the most upstream weir (Figure 10.5). Water temperature was available for Rices Weir up until January 2013 (Figure 10.5), this steadily increased throughout the study period; this was also used in the Bayesian analysis.

Environmental water was delivered from mid-August 2012 through to mid-May 2013 (Figure 10.5). An environmental flow of 40 ML/d was sent from mid-August 2012, which was increased to 250 ML/d until mid-January, when it was reduced to 200 ML/d, persisting until the end of the study period. The environmental flow was later reduced to 150 ML/d in mid-April, and persisted until mid-May where environmental flows ceased.

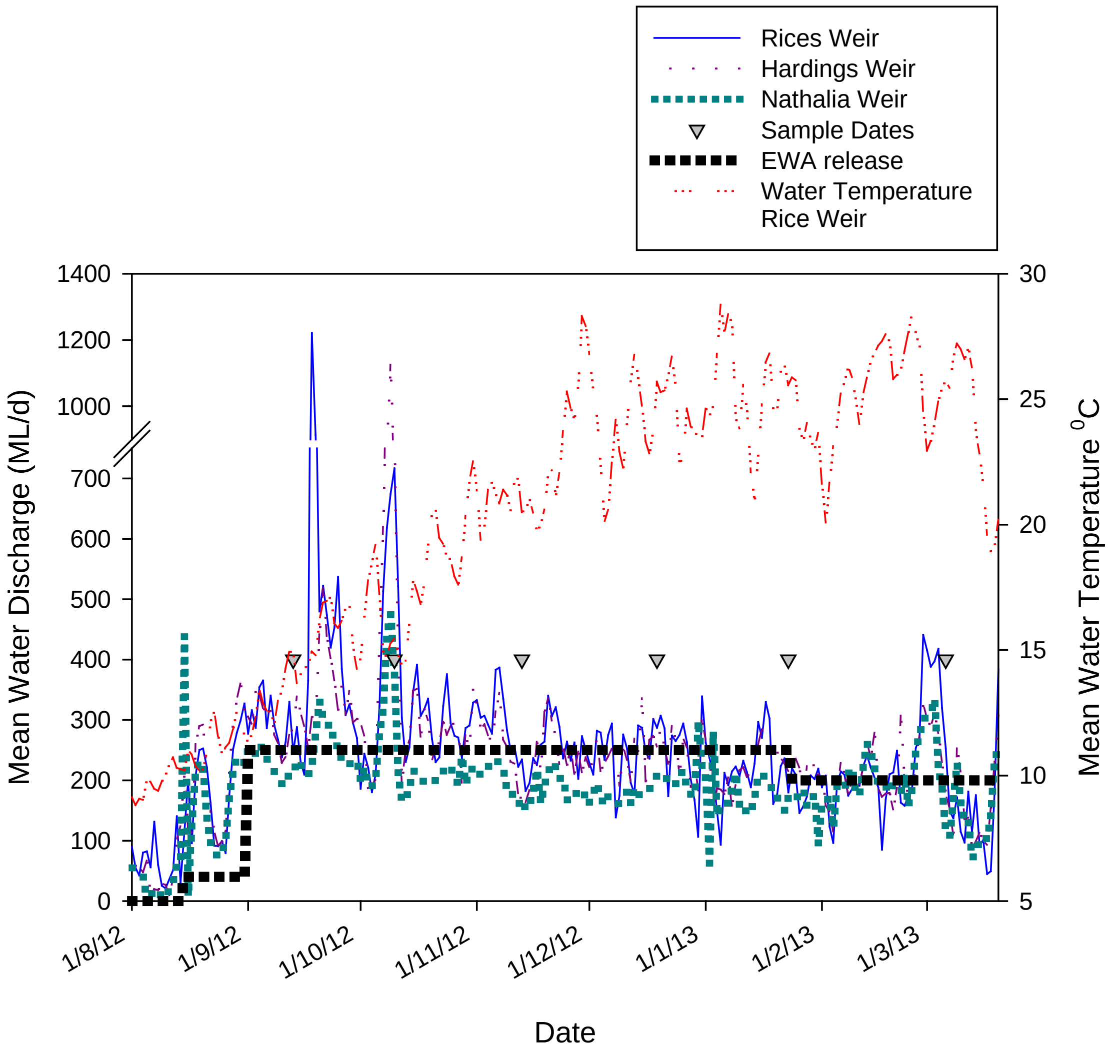


Figure 10.5: Fishway sample dates (arrows) and average discharge (ML/d) at three weirs along the Broken Creek. Note, only three weirs have been displayed for ease of interpretation. Average water temperature (^{0}C) is also shown for Rices Weir.

Fishway Trapping

A total of 125 fish, represented by five species, were captured from fishway trapping (Table 10.2). Common carp were the most common species sampled, followed by golden perch, Murray cod, silver perch and goldfish. Carp were collected in reasonable numbers during all sampling trips with the exception of September and January, Murray cod were collected in greatest numbers during October, while golden perch were collected during October and March (Table 10.2).

The smallest fish captured during sampling was a common carp at 241 mm long, while the largest was a Murray cod at 710 mm (Table 10.3). A length-frequency histogram of the most frequently sampled species, common carp, showed a range in size from 241 mm to 605 mm with the most common size of individuals between 340 – 400 mm (Figure 10.6).

Table 10.2 Total number of fish collected in fishway traps (all eight) from September 2012 to March 2013.

Trip	Sept	Oct	Nov	Dec	Jan	Mar	Total
Species							
Common carp	3	21	19	16	6	22	**87**
Goldfish	0	1	0	0	0	0	**1**
Silver perch	0	0	3	1	0	3	**7**
Murray cod	2	6	0	0	0	0	**8**
Golden perch	0	8	2	2	2	8	**22**
Total	**5**	**36**	**24**	**19**	**8**	**33**	**125**

Table 10.3 Total number and length (mm) (caudal fork length for fork tailed species) of fish species trapped in the Broken Creek fishway traps from September 2012 to March 2013.

Species	Number	Av. Length	Min Length	Max Length
Common carp	87	401.4	241	605
Goldfish	1	210	-	-
Silver perch	7	326.4	263	386
Murray cod	8	635.5	419	710
Golden perch	21	426.2	279	566
Total	**124**			

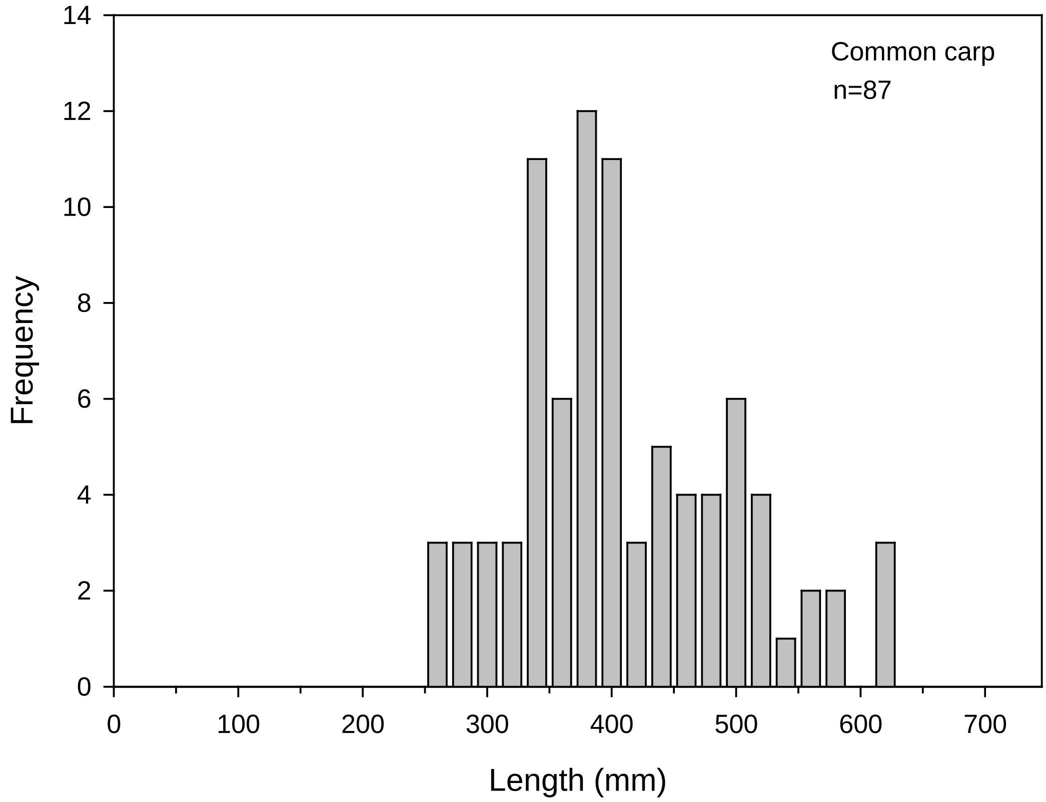


Figure 10.6: Length frequency of all common carp collected at fishways along Broken Creek.

Bayesian Analysis – Common carp

The number of common carp caught were highly variable with too many zeros and ones, and as such, negative binomial models were fitted (Figure 10-7, Table 10.4). Discharge and trip (or temporal variation) as fixed effects provided the lowest deviance information criterion (DIC) and therefore, the most evidence. This model fits the data relatively well, with the posterior predictive Bayesian *p*-value of 0.320, and only one Pearson residual is marginally greater than 4. The parameter estimates and 95% high density intervals (HDI) are shown below (Table 10.5). As none of the HDI exclude zero, there is insufficient evidence to claim that common carp use of the fishways is affected by either discharge or trip.

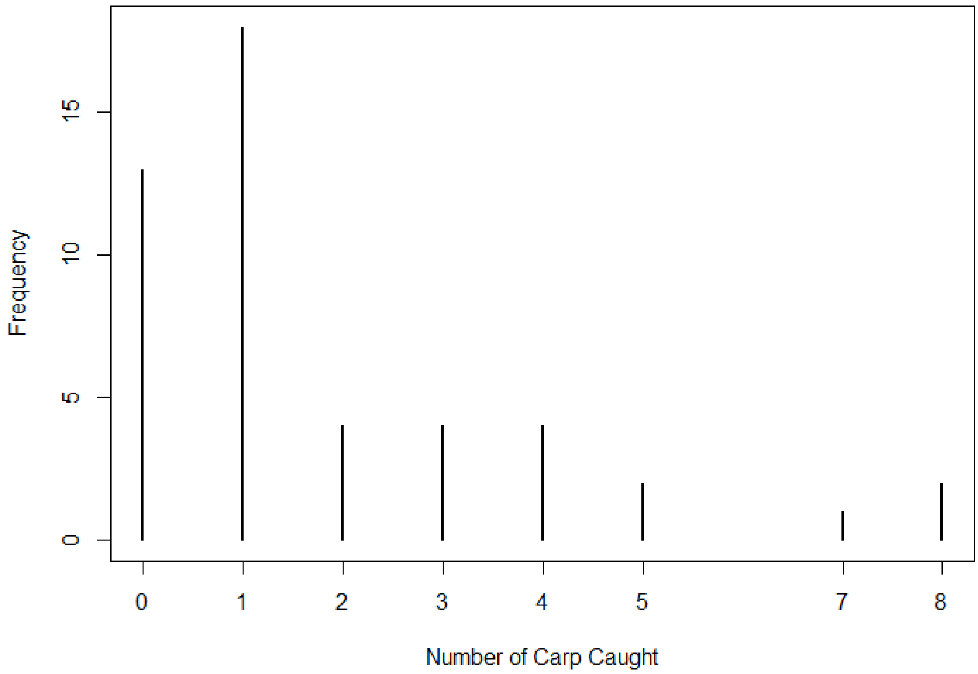


Figure 10-7: Plot of the frequency of the number of common carp caught per weir for each trip.

Table 10.4: The information criteria for each of the negative binomial models used for common carp capture numbers.

Model	DIC	ΔDIC
Discharge + Trip	178.6	
Discharge + Trip + Weir (Random)	180.3	1.7
Discharge + Temp	180.6	2.0
Discharge + Temp + Weir (Random)	182.1	3.5
Discharge + Trip + Weir (Fixed)	187.6	9.0
Discharge + Temp + Weir (Fixed)	189.3	10.7

Table 10.5: Parameter estimates (log scale, except for Size) and 95% high density intervals (HDI) for the negative binomial model for the number of common carp given the trip number and the flow.

Parameter	Estimate	95% HDI	
		Lower	Upper
Intercept	0.04	-0.65	0.77
Discharge	0.37	-0.02	0.80
Trip	0.20	-0.01	0.44
Size	1.82	0.55	3.71

Golden Perch

The number of golden perch caught during each trip follows a Poisson distribution (Figure 10.8). Therefore, Poisson models were fitted for golden perch.

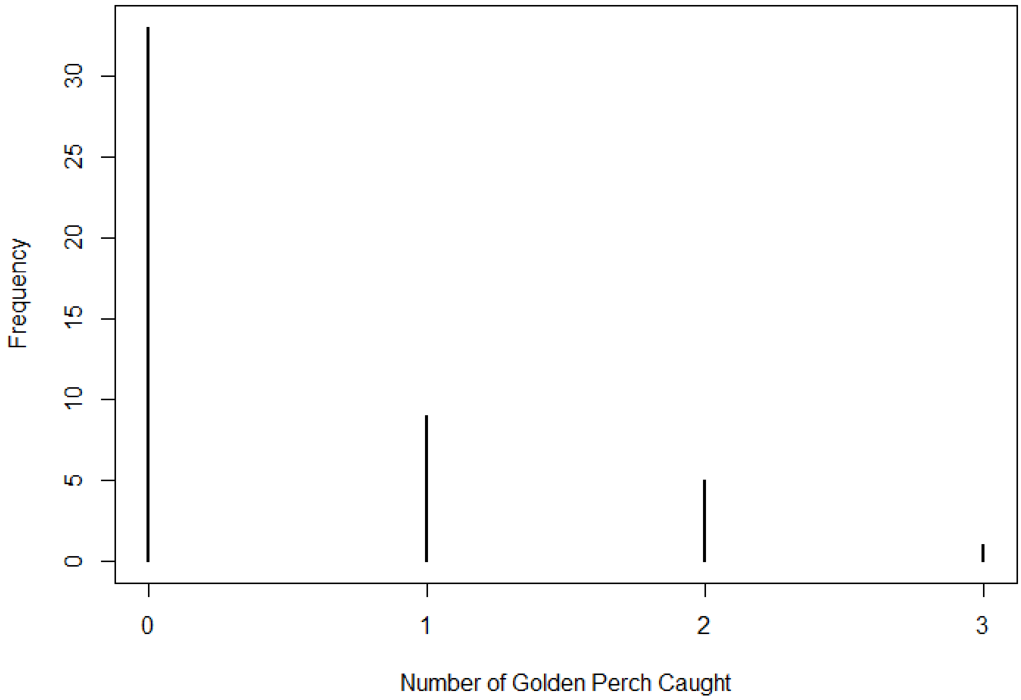


Figure 10.8: Plot of the frequency of the number of golden perch caught per weir for each trip.

The information criteria for each golden perch model is given in Table 10.6. The model with flow and trip as fixed effects provided the lowest DIC. This model fits the data relatively well with a posterior predictive Bayesian *p*-value of 0.240, and only one Pearson residual marginally greater than 4.

The parameter estimates and 95% high density intervals (HDI) are shown in Table 10.7. All of the HDI exclude zero, providing evidence to claim that golden perch use of the fishways is increased by both flow and over the survey period. The following equation describes how the parameter estimates in Table 10.7 fit the model.

Equation 1) $\ln(\hat{\mu}_{i,j}) = -2.34 + 0.7 \times Flow_{i,j} + 0.14 \times Trip_i$

Where the log of the estimated mean CPUE for golden perch (using the model with most evidence), and given standardised flow and the trip number.

Table 10.6: The information criteria for each of the Poisson models used for Golden Perch capture numbers.

Model	DIC	ΔDIC
Flow + Trip	83.1	
Flow + Trip + Weir (Random)	84.8	1.7
Flow + Temp	89.8	6.7
Flow + Temp + Weir (Random)	91.3	8.2
Flow + Trip + Weir (Fixed)	93.4	10.3
Flow + Temp + Weir (Fixed)	101.6	18.5

Table 10.7: Parameter estimates (log scale) and 95% high density intervals (HDI) for the Poisson model for the number of Golden Perch given the trip number and the flow.

Variable	Estimate	95% HDI	
		Lower	Upper
Intercept	-2.34	-3.62	-1.10
Flow	0.70	0.29	1.13
Trip	0.48	0.14	0.84

Murray cod

Murray cod were caught in low numbers (i.e. presence/absence) and as such, binomial (logistic) models were used for the analysis. The information criteria used for each model is shown below (Table 10.8), with the model with flow and temperature as fixed effects providing the lowest DIC i.e. most evidence. This model fits the data relatively well as the posterior predictive Bayesian *p*-value of 0.452 and no Pearson residuals greater than 4.

The parameter estimates and 95% high density intervals (HDI) for the best model is given in

Table 10.9. All of the HDI exclude zero, there is evidence to claim that Murray cod use of the fishways is increased by either increased flow or lower temperature. The following equation describes how the parameter estimates in

Table 10.9 fit the model.

Equation 2) $\ln\left(\frac{\hat{p}_{i,j}}{1-\hat{p}_{i,j}}\right) = -10.8 + 3.55 \times Flow_{i,j} - 6.51 \times Temp_i$

Where the logit of the estimated probability for Murray cod presence (using the model with most evidence), and given standardised flow and temperature.

Table 10.8: The information criteria for each of the binomial models used for Murray Cod capture numbers.

Model	DIC	ΔDIC
Flow + Temp	19.0	
Flow + Temp + Weir (Random)	20.5	1.5
Flow + Trip	21.4	2.4
Flow + Trip + Weir (Random)	23.0	4
Flow + Temp + Weir (Fixed)	27.9	8.9
Flow + Trip + Weir (Fixed)	29.8	10.8

Table 10.9: Parameter estimates (logit scale) and 95% high density intervals (HDI) for the binomial model for the number of Murray Cod given the trip number and the flow.

Variable	Estimate	95% HDI	
		Lower	Upper
Intercept	-10.80	-19.33	-3.31
Flow	3.55	1.27	6.12
Temperature	-6.51	-12.40	-0.73

PIT tagged fish

A total of 914 fish comprising three native fish species and three alien fish species have been implanted with PIT tags since 2005. Murray cod ranged in size from 83-1200 mm, with the most common size being tagged in the 280-320 mm range (Figure 10.9). Golden perch ranged in size from 40 – 580 mm, with the most frequent fish being tagged around 400 mm in length. Silver perch ranged in size from 263-286 mm (av 326.4 mm), while goldfish ranged in size from 185-320 (av. 413.3 mm), and redfin from 215-443 mm (av. 299 mm) (Figure 10.9). Common carp ranged in size from 152-672 mm, with the most frequent fish being tagged around 400 mm in length (Figure 10.9).

The number of PIT tagged fish recorded (per month) moving through the Broken Creek fishways was plotted against mean monthly water discharge and water temperature (Figure 10.10). The number of golden perch and Murray cod moving through the fishways appeared to increase with water discharge during the large flood event in 2010-11, while common carp movement was more variable (Figure 10.10). Water temperature did not appear to influence movement, instead following a cyclic pattern – increasing through the spring-summer period, and falling during the autumn-winter period.

Table 10.10: Details of fish PIT tagged between 2005 and 2013.

Lengths	*Common carp*	*Golden perch*	*Goldfish*	*Murray cod*	*Redfin perch*	*Silver perch*
Mean	412.2	395.0	239.0	469.8	298.8	326.4
Standard Error	4.9	6.4	16.1	11.7	16.6	14.3
Median	410.0	394.5	230.0	422.0	293.0	326.0
Mode	395.0	540.0	220.0	375.0	277.0	
Standard Deviation	100.8	92.5	42.6	187.6	59.8	37.9
Sample Variance	10164.1	8556.4	1812.0	35195.6	3574.7	1435.6
Kurtosis	-0.3	-0.8	2.0	1.4	1.7	1.2
Skewness	-0.1	-0.1	1.1	1.1	1.1	-0.2
Range	520.0	403.0	135.0	1117.0	228.0	123.0
Minimum	152.0	177.0	185.0	83.0	215.0	263.0
Maximum	672.0	580.0	320.0	1200.0	443.0	386.0
Sum	177254.0	82150.0	1673.0	121679.0	3884.0	2285.0
Count	430.0	208.0	7.0	259.0	13.0	7.0
Confidence Level(95.0%)	9.6	12.6	39.4	23.0	36.1	35.0

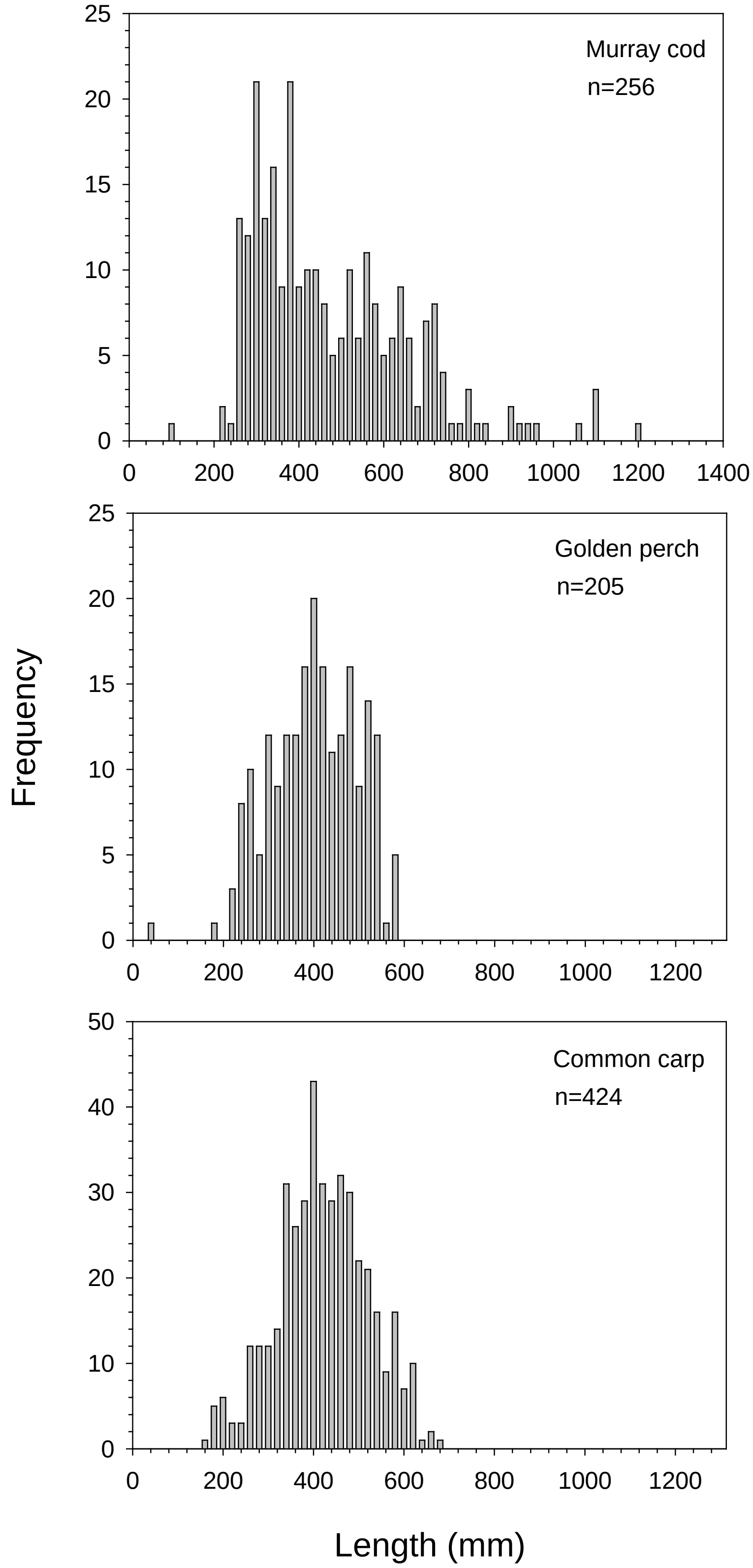


Figure 10.9: Length-frequency of fish PIT tagged in Broken Creek

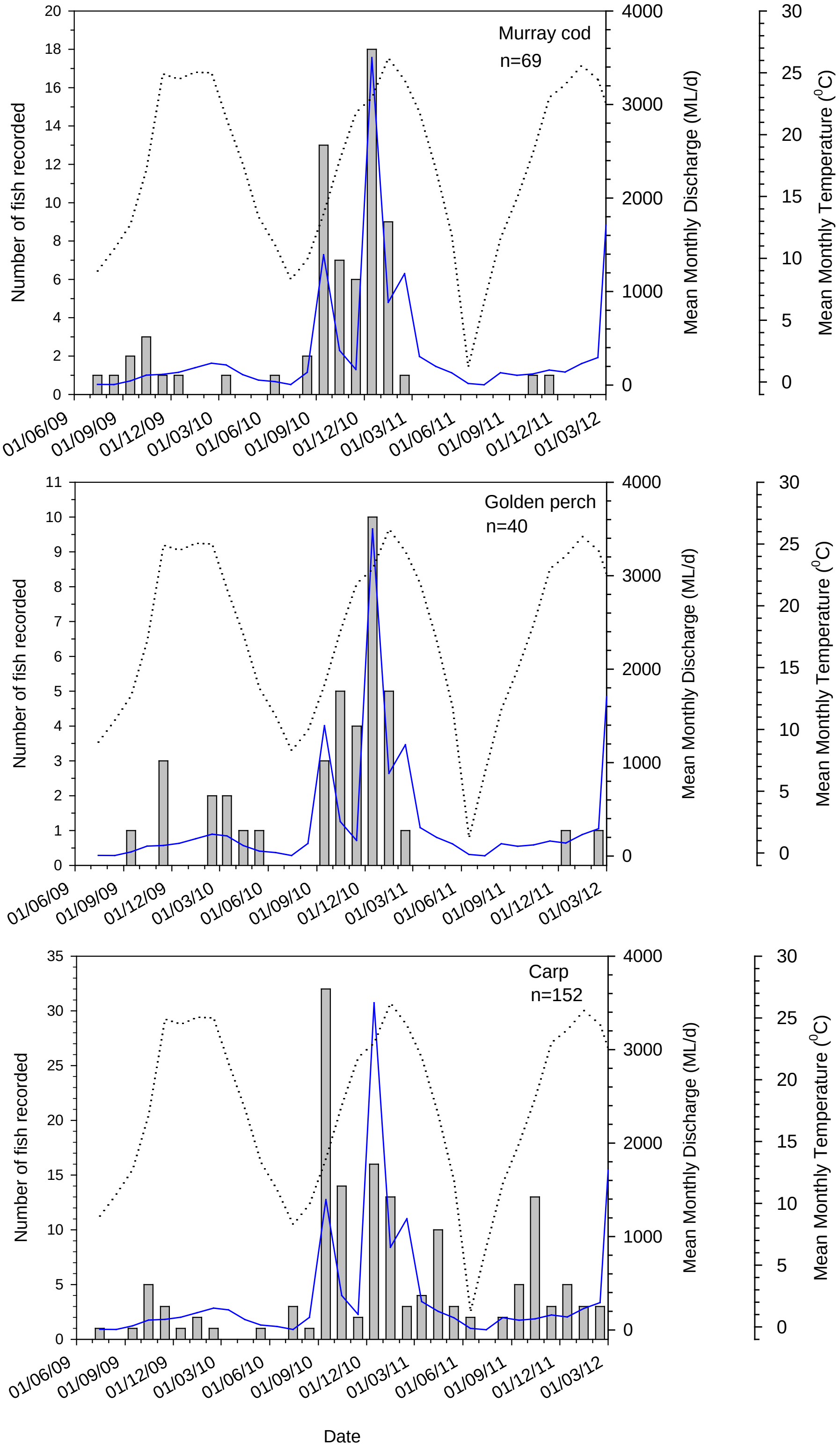


Figure 10.10: Number of PIT tagged fish recorded at fishways along Broken Creek per month. Solid line is Broken Creek discharge at Rices Weir, dotted line is water temperature at Rices Weir

10.4 Discussion

Common carp

The Bayesian binomial model used for common carp demonstrated that neither water discharge, water temperature, site (or weir), nor sample time (trip) were good predictors of common carp movement through fishways on Broken Creek. While the PIT tag movement data provided further evidence to support this. Water discharge and trip provided the model with the best fit, despite the HDI being less than zero indicating that there was no evidence to suggest that water discharge and trip were good predictors of common carp movement through fishways.

These data suggest, therefore, that common carp will move through fishways along Broken Creek regardless of time (Sept-March only) or water discharge, which is consistent with common carp movement data from the fishways along the Murray River e.g. (Mallen-Cooper, 1996). However, elevated flows will provide greater access to lateral habitats, such as low-lying floodplains, which common carp are known to prefer (Jones and Stuart, 2009). Common carp use low-lying floodplain habitats for spawning when temperatures exceed approximately 17^{o} C in early spring (Stuart and Jones, 2006a). Environmental watering strategies for Broken Creek should, therefore, consider that passage of common carp through the fishways is not likely to be influenced by creek discharge, but a spawning event may occur.

These findings should also be considered in the context of the relatively low flows that occurred during the 2012/13 irrigation season (August-May). Large flow events (including flooding) are likely to influence the behavior of common carp given that they are known to have flexible movement strategies and take advantage of suitable floodplain conditions when they become available (Jones and Stuart, 2009). Some common carp for example, are known to migrate more than 100 km to access floodplain habitats when they become available (Jones and Stuart, 2009, Stuart and Jones, 2006b).

Further, another investigation documented significant numbers of common carp juveniles drifting out of the Barmah-Millewa forest (BMF) floodplain and migrating up through the Rices Weir fishway (the lowest fishway on Broken Creek) in the 2000 flood event (Stuart and Jones, 2002). These data suggest that the BMF can be a significant source of common carp for Broken Creek, and a Broken Creek carp management/environmental watering plan should consider the broader context within the region.

Golden Perch

A Poisson Bayesian model including discharge (or flow) and trip provided the lowest DIC value (83.1) for golden perch, but unlike common carp where the 95% HDI were outside the error margin, golden perch showed a response, with increasing numbers of golden perch moving through fishways as discharge increased and as the season (or trips) progressed. These results are consistent with the PIT tag movement data, and data from previous investigations of golden perch which show that they are migratory/highly mobile (Reynolds, 1983, Jones, 2007, O'Connor et al., 2005), and particularly active between September and February/March (Mallen-Cooper, 1996, O'Connor et al., 2005).

Predictive data was generated for golden perch from Table 10.7 and Equation 1; predictive data allows waterway managers to determine the effect of changing various parameters on fish movement. This model suggests that with an average flow of 257.8 ML/d during September, 0.1 (0.03 to 0.34) golden perch are expected to pass through the Broken Creek fishways per day (i.e. 1 per ten days). However, at a higher discharge, i.e. 500 ML/d, the number of golden perch passing through the fishway increases to 0.29 /day (0.11 to 0.67) – a threefold increase (0.29 /day is equivalent 1 fish on average per 3.5 days). Further, if high discharges (500 ML/d) are delivered in January, then the number of fish passing through the fishway increases to 2.0 /day (0.85 to 4.83). Such results suggest that golden perch are closely linked with environmental cues.

The life-history of the target species should therefore, be considered when delivering water for the environment. If the aim of the environmental flow is for example, to stimulate a golden perch spawning event, then it is best done around October-November for the mid-Murray (King et al., 2008). However, if the aim is to stimulate upstream migration of golden perch, then later in the season (i.e. December-March) may provide better results (Mallen-Cooper, 1996). Similarly, golden perch undertake downstream movements in the Broken Creek, with higher flows resulting in more fish moving over Kennedys Weir (O'Connor et al., 2006). The same study also suggested that smoother flow delivery (i.e. weir approach velocity/turbulence) is likely to reduce behavioral inhibitions of fish moving downstream at instream structures. As such, delivering water in a fashion that facilitates downstream migration over the weirs will help to minimise the impact on instream barriers.

Silver Perch

In depth analysis was not undertaken on silver perch due to the low abundances collected throughout the investigation. The low abundances are however, consistent with previous findings on Broken Creek e.g. (O'Connor and Amtstaetter, 2008, O'Connor and O'Mahony, 2008). Similarly, the timing of fish movement (November, December and early March) documented during this investigation is also consistent with previous investigations at a fishway on the Murray River e.g. (Mallen-Cooper, 1996).

Providing environmental watering recommendations for silver perch based on this investigation is difficult and possibly erroneous. Managers of environmental water releases in Broken Creek should instead be guided by spawning and recruitment data (for the species) collected from the Murray River until data specific to the species in Broken Creek can be obtained. Silver perch are known to spawn from November to February and on within and overbank flows (Mallen-Cooper and Stuart, 2003, King et al., 2008), and a previous investigation along the Murray River found that silver perch increased their spawning activity (i.e. increased number of eggs/larvae detected) during the delivery of environmental water (King et al., 2009).

Murray cod

The results for Murray cod are not surprising given that they are most active in spring/early summer (Koehn et al., 2009a), and spawn in later October-December (Humphries, 2005). In fact, Murray cod are known to migrate large distances (<130 km) for spawning related activities, sometimes returning to the same site (Koehn et al., 2009a, Koehn and Harrington, 2006). These movement data also provide important records of Murray cod moving through fishways, as few studies have documented this.

Murray cod showed a decline in fishway usage as water temperatures increased/from September to March. However, this decline could be a due to the fact that Murray cod were only caught using the fishways in September and October, at the beginning of the surveys and when water temperatures were at their coldest (Figure 10.11). Unfortunately, there were no high flow events during the summer trapping months that could have helped identify if temperature was really a ‘driver’ for Murray cod movement through fishways. However, the PIT tag data suggests that Murray cod will move during the warmer months if flooding is occurring.

It is important to note that it is difficult to separate the effect of temperature and the trip (for Murray cod) given that there was a degree of collinearity between the two. Models for the other species also backed this up showing similar trends in the trip number and the temperature. Golden perch for example showed an increase in fishway usage from September to March, as temperature increased.

Predictive data was generated for Murray cod from Table 10.9 and Equation 2. These data suggest that with average flow (257.8ML) and average temperature (21.6°C) you would expect a 0.0% (0.0% to 3.6%) chance of Murray cod to use the fishway. However, if the flow was higher, 500ML and at an average temperature, then you would expect the chance of Murray cod using the fishway to increase to 0.5% (0.0% to 81.8%). Given average flow and the water temperature was 18°C, then the expected chance of Murray cod using the fishway is 0.3% (0.0% to 12.7%). Similarly, if there was a 500 ML flow and the temperature was 18°C (during spring), then the expected chance of catching a Murray cod is even greater at 43.6% (2.1% to 97.1%).

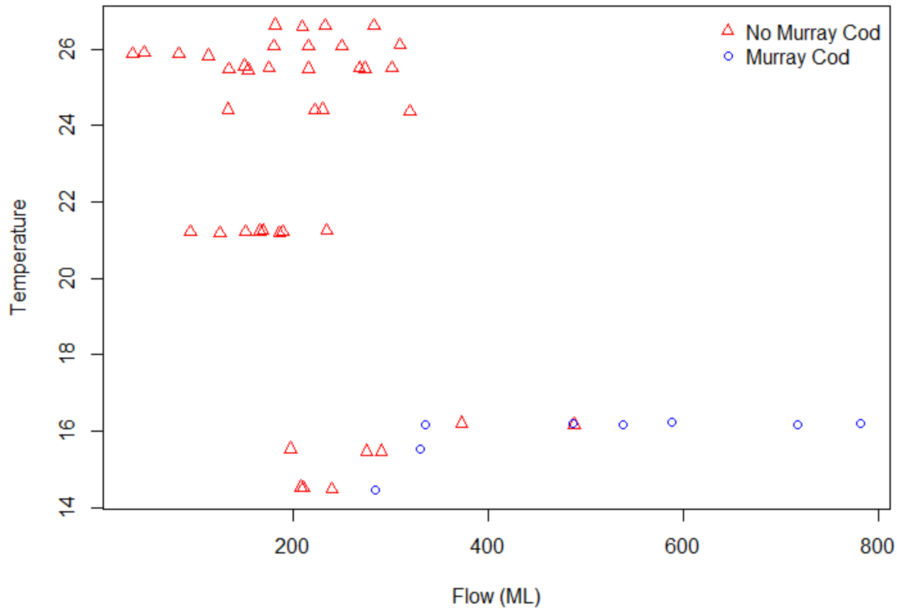


Figure 10.11: Plot of flow and temperature during the surveys differentiating when Murray cod were detected.

These data suggest that environmental water should be released early in the spring to stimulate Murray cod movement. Such information provides important insights into the behavior of the species, and as such, enables waterway managers to target pre-spawning migrations with environmental water, rather than trying to provide flows to cue spawning activities.

Environmental Flow Delivery

Streamflow, generated mostly through environmental releases, was generally stable for a large part of the study period (i.e. 250 ML/d during September 2012 to 24 January 2013), and importantly during the known spawning and migration season (October-February) of large-bodied native fish e.g. (King et al., 2008, King et al., 2009, Humphries et al., 2002).

A recent review of 165 scientific papers on flow alterations and associated ecological response found that stabilization of the magnitude and frequency of flows generally had a detrimental effect on aquatic and riparian species (Poff and Zimmerman, 2010). Specifically, aquatic species had a loss of species sensitivity, reduced diversity and abundances, increases in non-native species, reduced habitat for young fishes, and altered assemblages and dominant taxa.

Delivering environmental water in the Broken Creek should therefore, include daily and weekly variability where possible, in addition to flushing flows. Native fish are cued to variable flows or river rises (Mallen-Cooper, 1996, Jones, 2009, Jones, 2007), which facilitates upstream migration for feeding, spawning, dispersal or colonization purposes.

The maximum artificial discharge able to be sent through the Broken Creek system is 250-350 ML/d (Geoff Earl, GBCMA, pers. Comm.). This discharge is quite low and as such, daily, weekly, and monthly variations in flow may become more important for fish in Broken Creek, as has been found in the Murray River (Mallen-Cooper, 1996). Native fish responded quickly to short-term (<1 week) flushing flows during this investigation (M. Jones pers. comm.) and as such provision of a steady within channel rise over 1-2 weeks may induce a similar movement response. Similarly, environmental water could be used in conjunction with natural flow events to heighten peak flows and/or extend the duration of the flow event.

Separating movement in relation to creek discharge from the movement attributed to environmental water is difficult as one is typically coupled with the other. Having zero flows within a creek and then providing flows of different duration and size is a way to determine the cause and effect of environmental water on fish movement, however this is rarely achievable in natural systems. Another possibility is to repeatedly sample fish movement over an extended period (i.e. years) to maximise the chance of encountering various flow scenarios and baseline flow conditions. With this in mind, sampling the Broken Creek fishways has provided insightful information into the effect of environmental water on fish movement, however it is recommended that the investigation be continued for the 2013/14 period to increase the certainty associated with the findings.

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APPENDIX A: SPECIES LISTS FOR THE VEGETATION SURVEYS

Table A.1 Complete species list, with presence/absence for each site on lower Broken Creek.

Species	*Rices Weir*	*Kennedys Weir*	*Schiers Weir*	*Balls Weir*
Acetosella vulgaris	X	X	X	X
Ailanthus altissima		X		
Alopecurus sp.	X			
Alternanthera denticulata	X			
Asperula scoparia				X
Aster subulatus	X	X	X	X
Atriplex semibaccata	X		X	
Austostipa nodosa		X	X	X
Austrodanthonia sp.	X	X	X	X
Avena spp.	X	X		X
Bromus catharticus		X	X	X
Bromus diandrus	X	X	X	X
Calotis scapigera	X			
Carduus tenuiflorus				X
Carex gaudichaudiana		X	X	X
Carex inversa	X	X	X	X
Carex tereticaulis		X	X	X
Centipeda cunninghamii	X	X		X
Centipeda minima	X	X	X	X
Cerastium glomeratum	X			
Chondrilla juncea		X		
Cirsium vulgare	X	X	X	X
Conringia orientalis			X	
Conyza bonariensis	X	X	X	X
Cotula australis	X			
Crassula helmsii	X			
Cynodon dactylon	X	X	X	X
Cyperus eragrostis	X	X	X	X
Dactylis glomerata				X
Eclipta platyglossa	X			
Ehrharta longiflora			X	
Eleocharis acuta	X			
Elymus scaber				X
Epilobium billardierianum	X			
Epilobium hirsutum	X	X	X	X
Eucalyptus camaldulensis	X	X	X	X
Euchiton sphaericus	X	X	X	X
Euphorbia dallachyana		X	X	
Euphorbia drummondii	X			
Fraxinus angustifolia angustifolia			X	

Species	*Rices Weir*	*Kennedys Weir*	*Schiers Weir*	*Balls Weir*
Gamochaeta americana		X		
Glyceria declinata	X	X		
Haloragis aspera	X			X
Helminthotheca echioides	X			
Hermartia uncinata	X			
Hordeum geniculatum			X	
Hydrocotyle sibthorpioides				X
Hypochaeris radicata	X	X	X	X
Juncus amabilis				X
Juncus articulatus	X			
Juncus bufonius		X		
Juncus usitatus	X	X	X	X
Lachnagrostis filiformis	X	X	X	X
Lactuca serriola	X	X	X	X
Lepidium africanum			X	
Lolium perenne	X	X	X	X
Long thin leaf daisy	X			
Ludwigia peploides	X	X	X	
Lythrum hyssopifolia	X	X		
Malva parviflora			X	
Marrubium vulgare		X		
Marsilea drummondii	X			
Medicago polymorpha	X	X	X	
Modiola caroliniana		X		
Muehlenbeckia florulenta		X		
Oxalis exilis	X		X	
Panicum capillare	X			
Paspalum dilatatum			X	X
Paspalum distichum	X	X	X	X
Persicaria decipiens	X	X		
Persicaria hydropiper	X	X	X	X
Phalaris paradoxa				X
Plantago lanceolata				X
Polygonum aviculare	X		X	X
Polypogon monspeliensis	X			
Pseudognaphalium luteoalbum	X	X	X	X
Ranunculus sceleratus	X		X	X
Romulea rosea	X	X		X
Rosa rubiginosa		X		
Rumex brownii	X	X	X	X
Rumex conglomeratus	X		X	
Senecio linearifolius	X			
Senecio quadridentatus	X	X	X	X
Solanum elaeagnifolium	X		X	

Species	*Rices Weir*	*Kennedys Weir*	*Schiers Weir*	*Balls Weir*
Solanum nigrum				X
Sonchus asper	X	X	X	
Sonchus oleraceus	X	X	X	X
Spergularia sp.	X			
Subshrub sp	X			
Teucrium racemosum		X		
Trifolium angustifolium		X	X	X
Trifolium arvense	X	X	X	
Trifolium campestre	X			
Trifolium repens				X
Triglochin procera	X			
Typha orientalis		X		
Unknown	X		X	
Unknown forb no. 1	X			
Unknown forb no. 3				X
Unknown forb no. 4			X	
Unknown forb no. 5	X			
Unknown forbe no. 2				X
Unknown pasture grass		X		
Vittadinia cervicularis		X	X	
Vittadinia gracilis	X	X		
Vulpia bromoides	X	X	X	
Wahlenbergia flumenalis	X			

Table A.2: Complete species list, with presence/absence for each site on the Goulburn River.

Species	*Moss Rd*	*Darcy Tk*	*Loch Garry*	*McCoys Bridge*
Acacia dealbata	X	X	X	X
Acacia spp.	X		X	
Acaena novae-zelandeae	X			
Acetosella vulgaris	X	X		X
Aira elegantissima	X	X		
Alternanthera denticulata	X	X	X	X
Alternanthera denticulata s.l.	X	X	X	X
Anagallis arvensis	X	X		
Arctotheca calendula	X	X	X	X
Asperula conferta			X	
Asperula scoparia		X		
Asperula spp.		X		
Aster subulatus		X	X	X
Asteraceae spp.				X
Atriplex semibaccata				X
Austrodanthonia caespitosa	X	X		X
Austrodanthonia duttoniana				X
Austrodanthonia racemosa	X			X
Austrodanthonia racemosa var. *racemosa*	X	X	X	
Austrodanthonia setacea			X	X
Austrodanthonia spp.	X	X	X	X
Austrostipa spp.	X			X
Avena barbata	X		X	X
Avena fatua	X	X	X	X
Avena spp.		X	X	X
Bassia quinquecuspis				X
Brachypodium distachyon	X			
Brachyscome basaltica var. *gracilis*			X	X
Brassica sp.	X			
Briza maxima		X		
Bromus catharticus	X	X	X	X
Bromus diandrus	X	X	X	X
Bromus hordeaceus		X	X	X
Bromus hordeaceus subsp.		X		
Callistemon sieberi	X	X	X	
Callitriche sonderi		X		
Calocephalus sonderi			X	
Calotis scapigera				X
Capsella bursa-pastoris	X			
Cardamine gunnii			X	
Carduus pycnocephalus	X	X	X	X
Carduus spp.	X			

Species	*Moss Rd*	*Darcy Tk*	*Loch Garry*	*McCoys Bridge*
Carduus tenuiflorus		X		
Carex appressa	X	X	X	X
Carex bichenoviana	X	X	X	X
Carex breviculmis		X	X	
Carex inversa		X	X	X
Carex sp.			X	
Carex spp.		X	X	X
Carex tereticaulis	X	X	X	X
Caryophyllaceae sp.			X	
Cassinia arcuata	X			
Centipeda cunninghamii		X	X	X
Centipeda minima s.l.	X			
Cerastium spp.		X	X	X
Characeae spp.		X		
Chenopodiaceae spp.			X	
Chloris sp.				X
Chloris truncata				X
Cirsium vulgare	X	X	X	X
Convolvulus erubescens spp. agg.				X
Conyza bonariensis	X	X	X	X
Conyza spp.		X	X	X
Cotula coronopifolia	X			
Crassula decumbens var. *decumbens*	X		X	X
Crassula helmsii	X	X		X
Crassula sieberiana s.l.		X		
Crataegus monogyna		X		
Crepis capillaris				X
Cynodon dactylon	X			X
Cynodon dactylon var. *dactylon*	X	X	X	X
Cynodon dactylon var. *pulchellus*	X			X
Cyperus eragrostis	X	X		X
Cyperus exaltatus	X	X	X	X
Cyperus gunnii subsp. *gunnii*			X	
Cyperus sp.			X	X
Dactylis glomerata	X	X		
Dianella sp. aff. *longifolia* (Riverina)		X	X	
Dichondra repens	X	X		
Dillwynia cinerascens s.l.		X		
Echium plantagineum				X
Ehrharta erecta var. *erecta*	X			
Ehrharta longiflora	X	X	X	X
Einadia nutans subsp. *nutans*			X	X
Elatine gratioloides				X
Elymus multiflorus	X	X	X	X
Elymus scaber var. *scaber*			X	
Enteropogon acicularis				X

Species	*Moss Rd*	*Darcy Tk*	*Loch Garry*	*McCoys Bridge*
Epilobium billardierianum	X			X
Eragrostis spp.	X		X	X
Eryngium ovinum		X		X
Eucalyptus camaldulensis	X	X	X	X
Eucalyptus microcarpa	X		X	
Euchiton involucratus s.l.	X			
Euchiton sp.				X
Fumaria spp.		X		
Galium aparine	X	X	X	X
Gamochaeta purpurea s.l.				X
Geranium retrorsum s.l.		X		
Geranium sp.		X		
Geranium sp.	X	X	X	
Geranium sp. 2			X	
Geranium spp.		X		
Gonocarpus tetragynus		X		
Haloragis heterophylla				X
Helichrysum luteoalbum	X	X	X	
Helichrysum rutidolepis s.l.				X
Heliotropium europaeum	X			X
Helminthotheca echioides			X	
Hemarthria uncinata var. *uncinata*	X	X	X	X
Holcus lanatus	X			
Hordeum glaucum	X			
Hordeum leporinum	X	X	X	X
Hordeum marinum	X			
Hordeum murinum s.l.				X
Hordeum sp.				X
Hordeum spp.			X	
Hordium vulgare	X			
Hypochoeris glabra	X	X		
Hypochoeris radicata	X	X	X	X
Isolepis cernua	X			
Isolepis inundata	X		X	
Joycea pallida				X
Juncus amabilis	X	X	X	X
Juncus aridicola		X		X
Juncus bufonius	X			
Juncus falcatus			X	
Juncus flavidus		X	X	X
Juncus pallidus		X		
Juncus sp.	X		X	X
Juncus spp	X			X
Juncus spp.	X	X		X
Juncus spp.			X	
Juncus usitatus	X	X	X	X

Species	*Moss Rd*	*Darcy Tk*	*Loch Garry*	*McCoys Bridge*
Lachnagrostis filiformis var. 1	X	X	X	X
Lachnagrostis filiformis var. 2		X	X	
Lactuca sativa		X		X
Lactuca serriola	X	X	X	X
Lamium	X			
Lamium amplexicaule	X		X	
Leontodon taraxacoides		X		
Leontodon taraxacoides subsp. *taraxacoides*	X	X		X
Lepidium africanum	X			X
Lepidium spp.			X	
Leptospermum obovatum	X	X		
Linum marginale		X	X	
Lobelia concolor				X
Lobelia spp.		X		
Lolium perenne	X	X	X	
Lolium perenne var. perenne	X	X	X	X
Lolium rigidum		X	X	
Lolium spp.			X	
Ludwigia peploides subsp. montevidensis				X
Ludwigia spp.	X			
Lythrum hyssopifolia	X	X	X	X
Malva parviflora	X			
Malva parvifolium	X			
Malva spp.	X			
Marrubium vulgare				X
Medicago polymorpha	X	X	X	X
Mentha australis	X	X	X	X
Microlaena stipoides var. *stipoides*	X	X		
Modiola caroliniana		X	X	X
Muellerina eucalyptoides		X		
Myriophyllum aquaticum	X			
Oxalis perennans	X	X	X	X
Oxalis pes-caprae	X			X
Panicum coloratum		X	X	X
Panicum spp.				X
Paspalidium jubiflorum		X	X	X
Paspalum dilatatum	X	X		X
Paspalum distichum	X	X	X	X
Persicaria decipiens		X	X	
Persicaria hydropiper	X	X	X	X
Persicaria prostrata	X	X	X	X
Persicaria spp.			X	
Persicaria subsessilis		X	X	X
Phalaris aquatica	X	X		
Phalaris minor				X
Phragmites australis	X	X		X

Species	*Moss Rd*	*Darcy Tk*	*Loch Garry*	*McCoys Bridge*
Phyla canescens				X
Plantago lanceolata	X	X		X
Poa annua			X	X
Poa labillardierei	X	X	X	X
Poa sieberiana			X	
Polycarpon tetraphyllum	X			
Polygonum aviculare	X			
Polygonum aviculare s.l.	X	X	X	X
Polypogon monspeliensis	X			
Potamogeton crispus				X
Prunella vulgaris		X		
Pycnosorus globosus		X		
Ranunculus inundatus		X		
Ranunculus lappaceus		X		
Ranunculus muricatus				X
Ranunculus plebeius		X		
Ranunculus repens			X	
Ranunculus sceleratus ssp. sceleratus			X	
Ranunculus sessiliflorus		X		
Romulea rosea	X	X	X	X
Rorippa palustris			X	X
Rosa rubiginosa	X	X		
Rubus parvifolius		X		
Rumex brownii	X	X	X	X
Rumex conglomeratus			X	X
Rumex crispus	X	X		X
Rumex pulcher	X			
Rumex spp.	X		X	
Rytidosperma caespitosum	X			X
Rytidosperma pallidum				X
Rytidosperma racemosum	X		X	X
Rytidosperma setaceum				X
Rytidosperma sp.	X	X	X	
Salix alba x *fragilis*				X
Senecio quadridentatus	X	X	X	X
Senecio sp.	X			X
Senecio spp.		X		
Sida corrugata				X
Sigesbeckia orientalis			X	
Silybum marianum			X	
Solanum americanum		X		
Solanum nigrum	X	X		X
Solanum nigrum s.s.	X	X		X
Solanum pseudocapsicum	X			
Solanum sp.			X	X
Sonchus asper		X		

Species	*Moss Rd*	*Darcy Tk*	*Loch Garry*	*McCoys Bridge*
Sonchus asper s.l.	X	X	X	X
Sonchus asper ssp*. glaucescens*	X	X	X	
Sonchus asper subsp. *glaucescens*	X	X	X	
Sonchus oleraceus	X	X	X	X
Stackhousia monogyna			X	
Stellaria media				X
Sysimbrium spp.	X			
Themeda australis		X		
Themeda triandra		X	X	
Trifolium angustifolium	X			
Trifolium angustifolium var. angustifolium		X	X	
Trifolium angustrifolium	X		X	X
Trifolium glomeratum			X	X
Trifolium repens	X			X
Trifolium sp.	X			
Trifolium spp.	X	X		X
Trifolium subterraneum	X			
Triglochin procera s.l.	X	X		
Triticum aestivum				X
Typha domingensis				X
Unknown	X			
Urtica urens		X		
Verbena officinalis		X	X	X
Verbena officinalis s.s.		X		
Verbena spp.			X	
Veronica gracilis	X	X	X	X
Vicia hirsuta		X		
Vicia sativa	X	X	X	
Vicia spp.		X		
Vulpia bromoides	X	X	X	X
Wahlenbergia communis			X	X
Wahlenbergia communis s.l.			X	
Wahlenbergia fluminalis			X	
Wahlenbergia gracilis		X		
Wahlenbergia sp.		X	X	X
Wahlenbergia spp.	X	X	X	X
Walwhalleya proluta				X

APPENDIX B: VEGETATION TRANSECTS ON THE LOWER BROKEN CREEK

N.B. Precise coordinates of the transect endpoints are available from the authors.

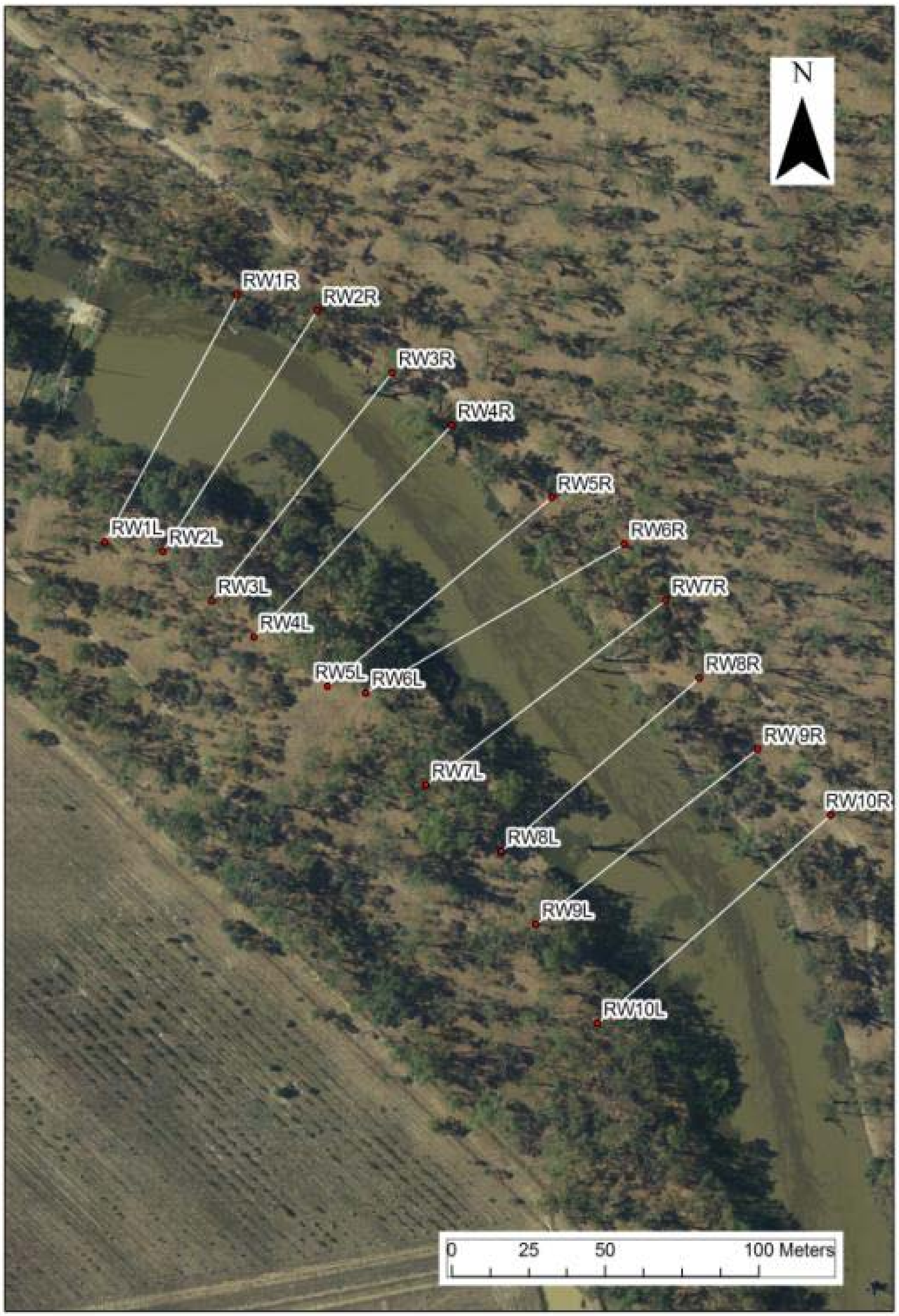

Figure B.1: Rices Weir

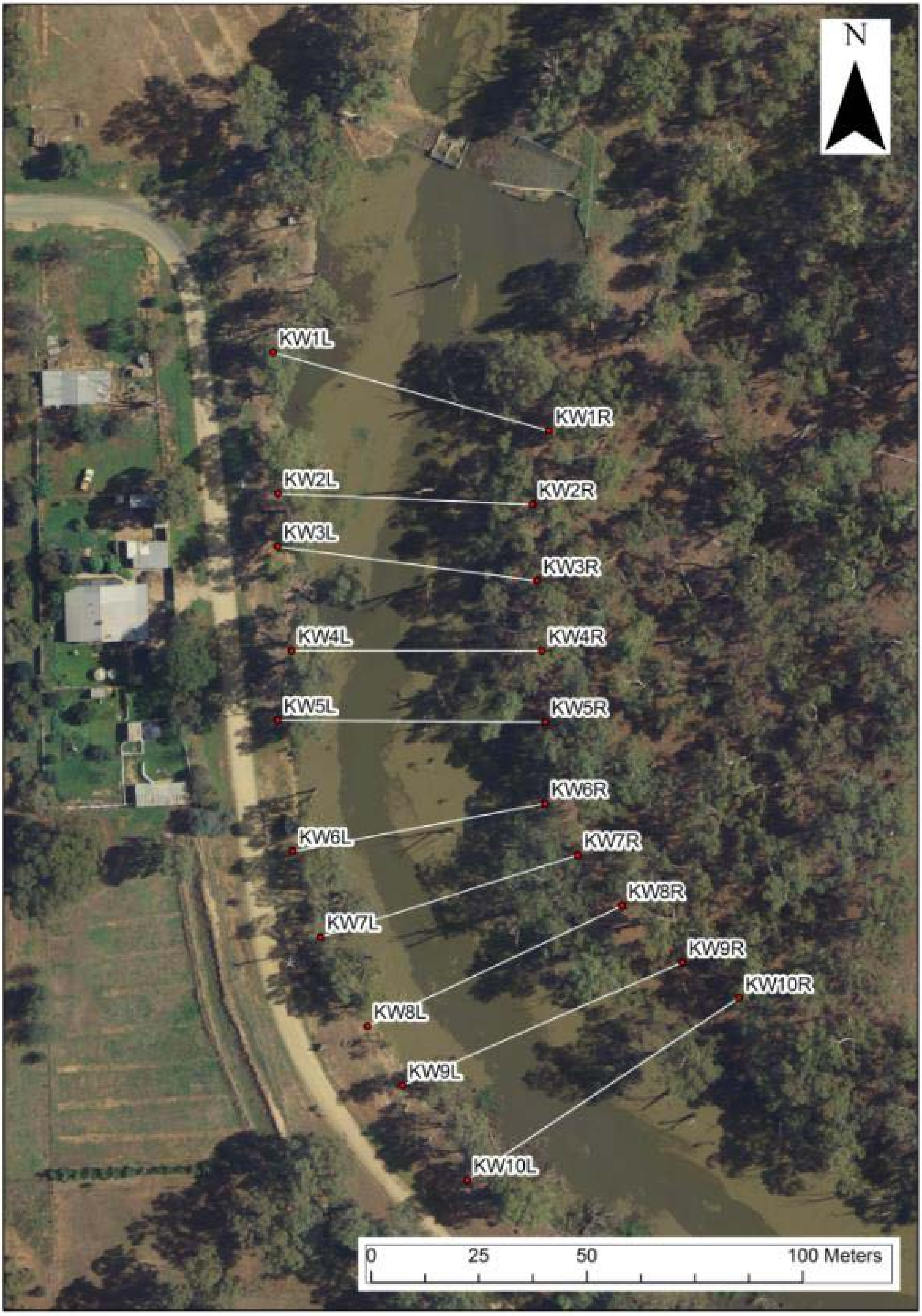

Figure B.2: Kennedys Weir.

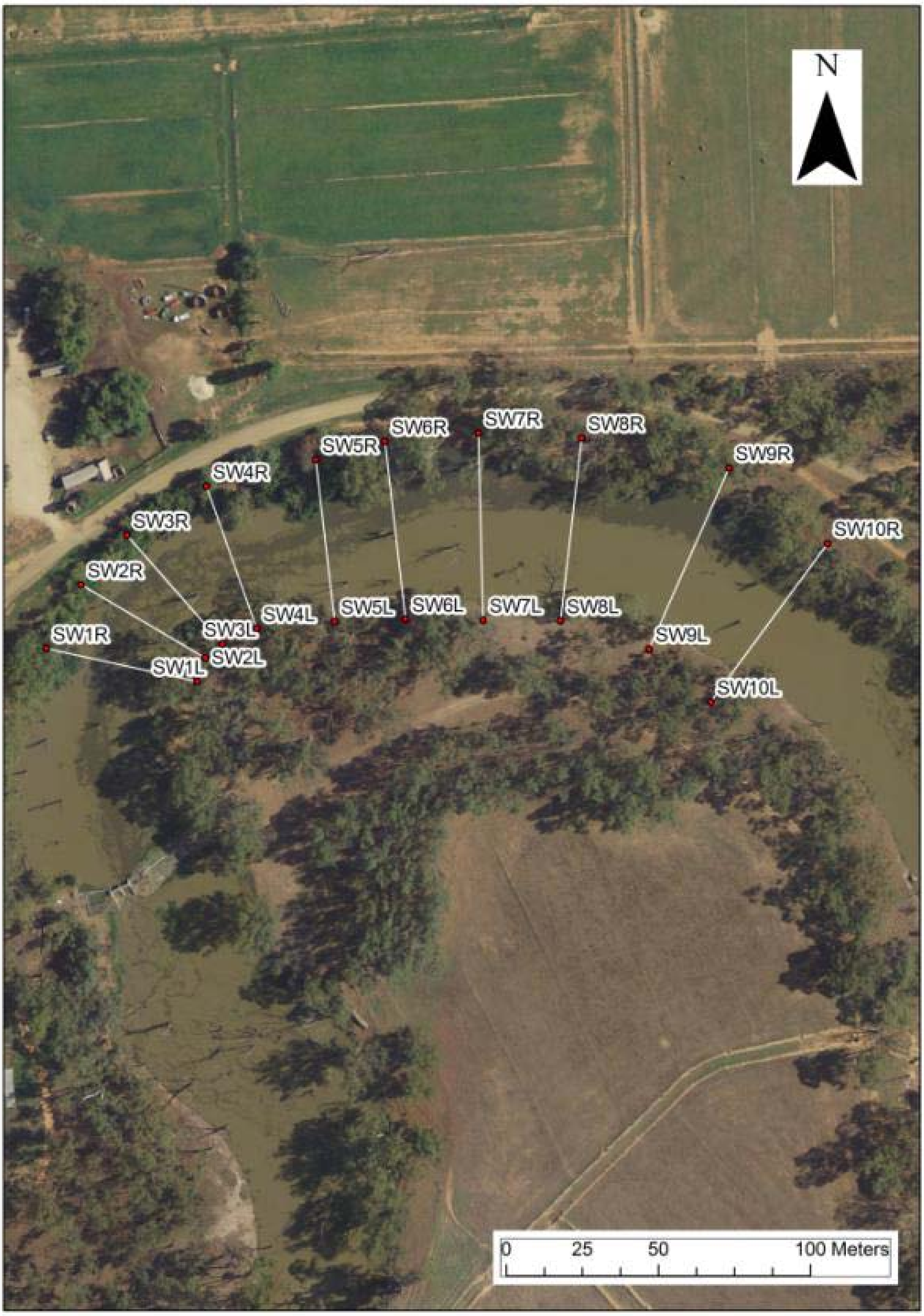

Figure B.3: Schiers Weir

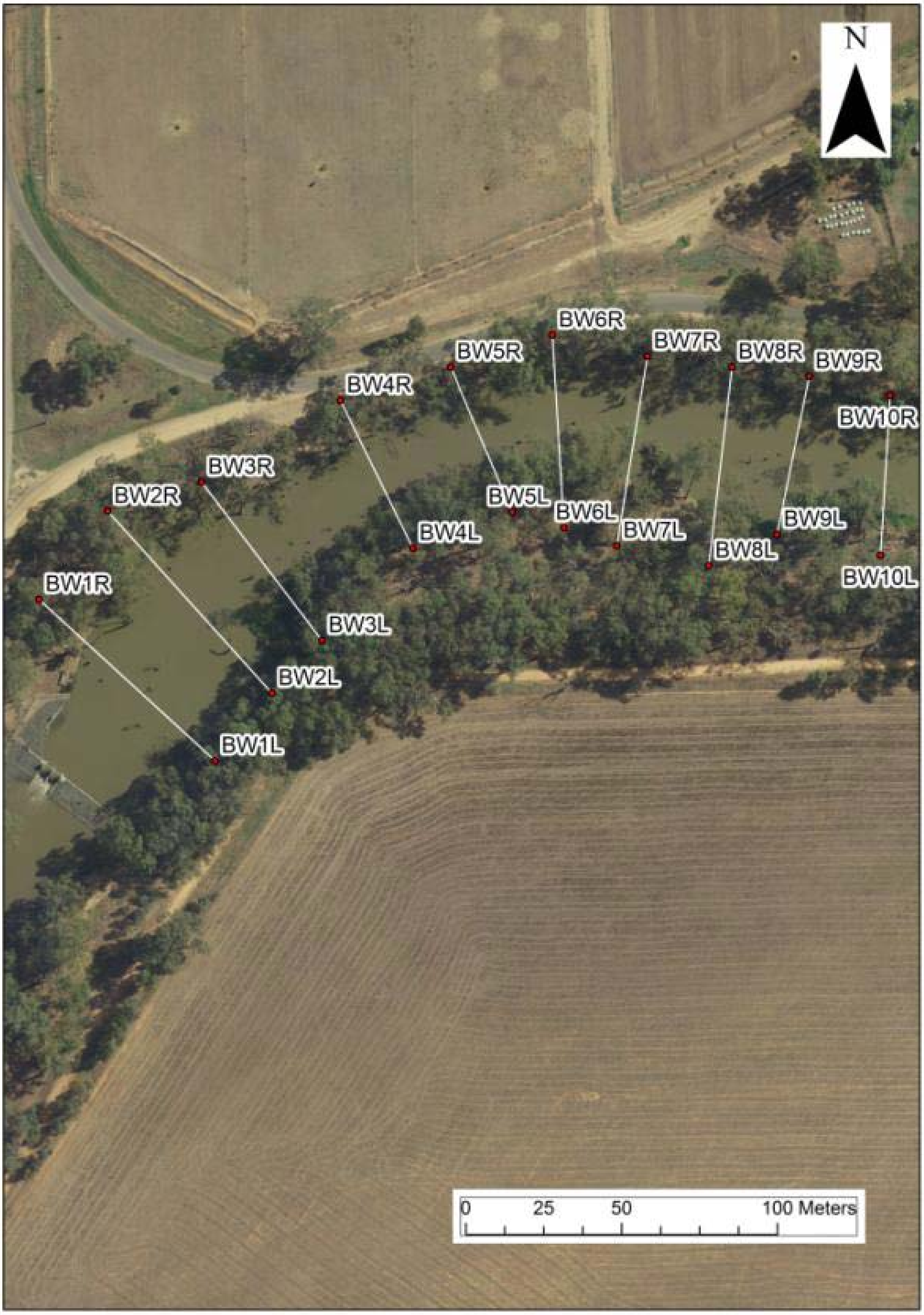

Figure B.4: Balls Weir.

APPENDIX C: EXPLORATORY DATA ANALYSIS PLOTS FROM THE LOWER GOULBURN RIVER VEGETATION ANALYSIS.

See next page.

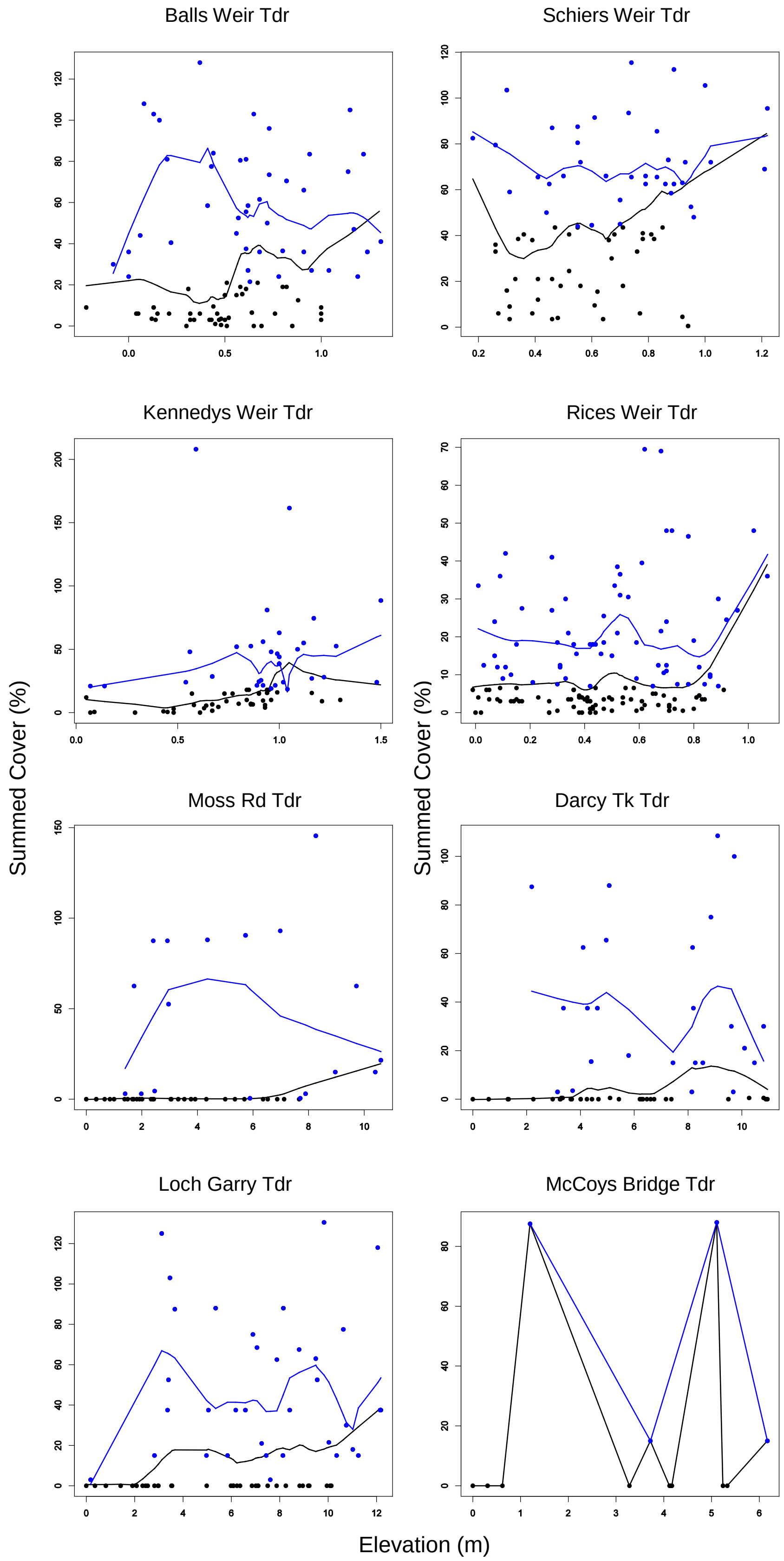

Figure C.1: Cover of Terrestrial dry species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Blue points are those in the top 50% of abundances. Lines are LOWESS smoothers for the whole data set (black) and the top 50% of data (blue).

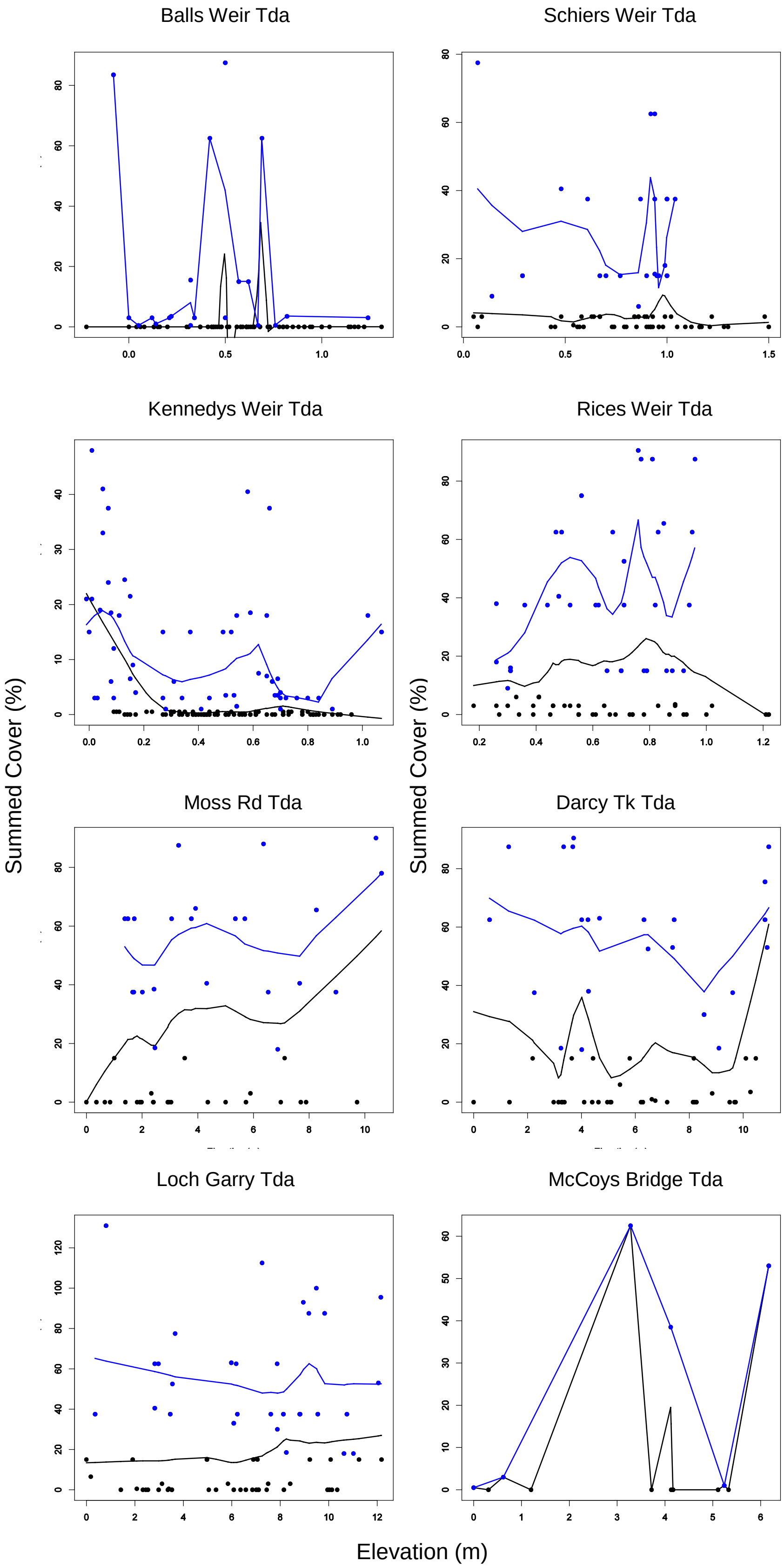

Figure C.2: Cover of Terrestrial damp species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Blue points are those in the top 50% of abundances. Lines are LOWESS smoothers for the whole data set (black) and the top 50% of data (blue).

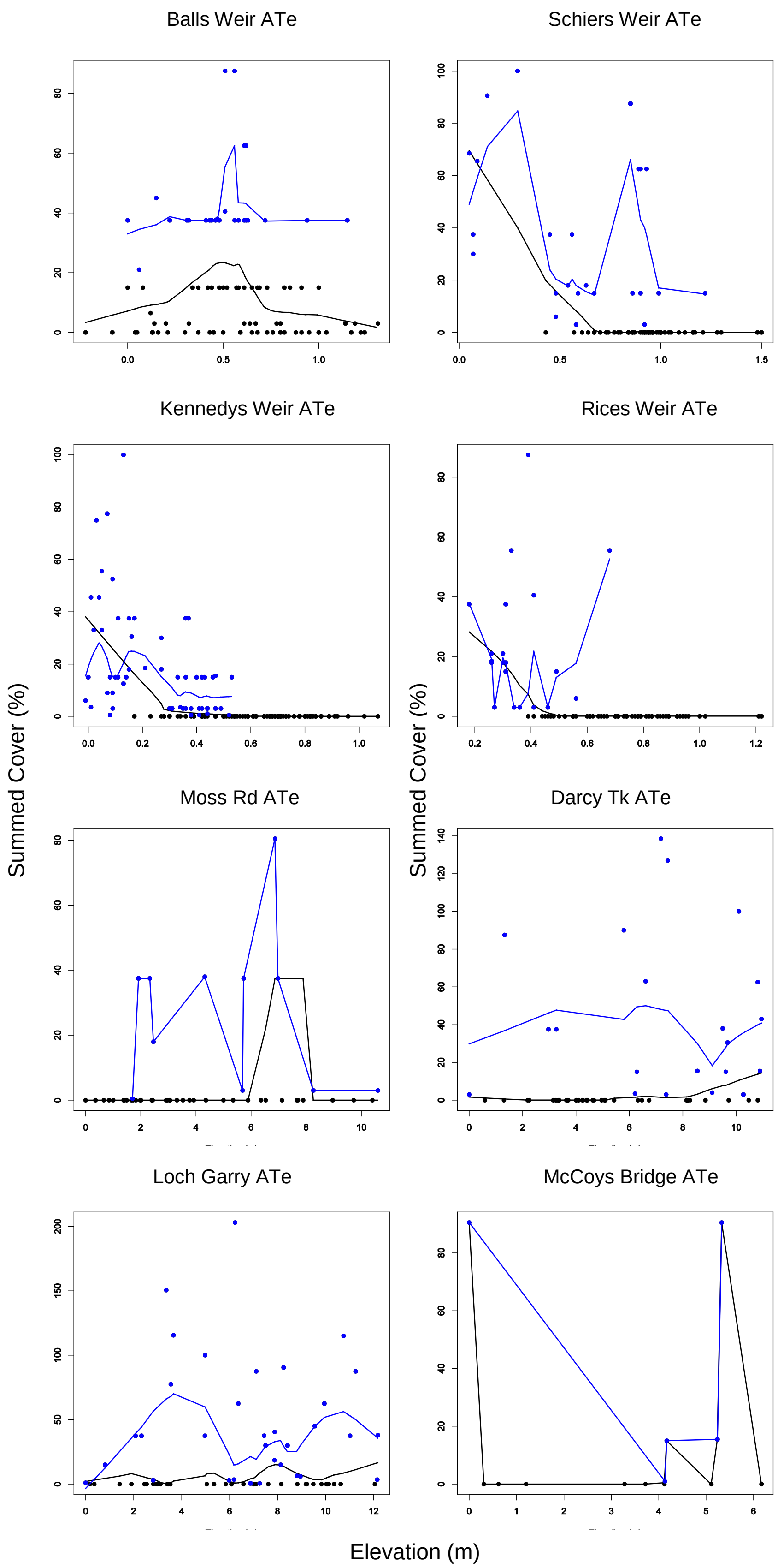

Figure C.3: Cover of Amphibious fluctuation tolerator, emergent species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Points and lines are as described above.

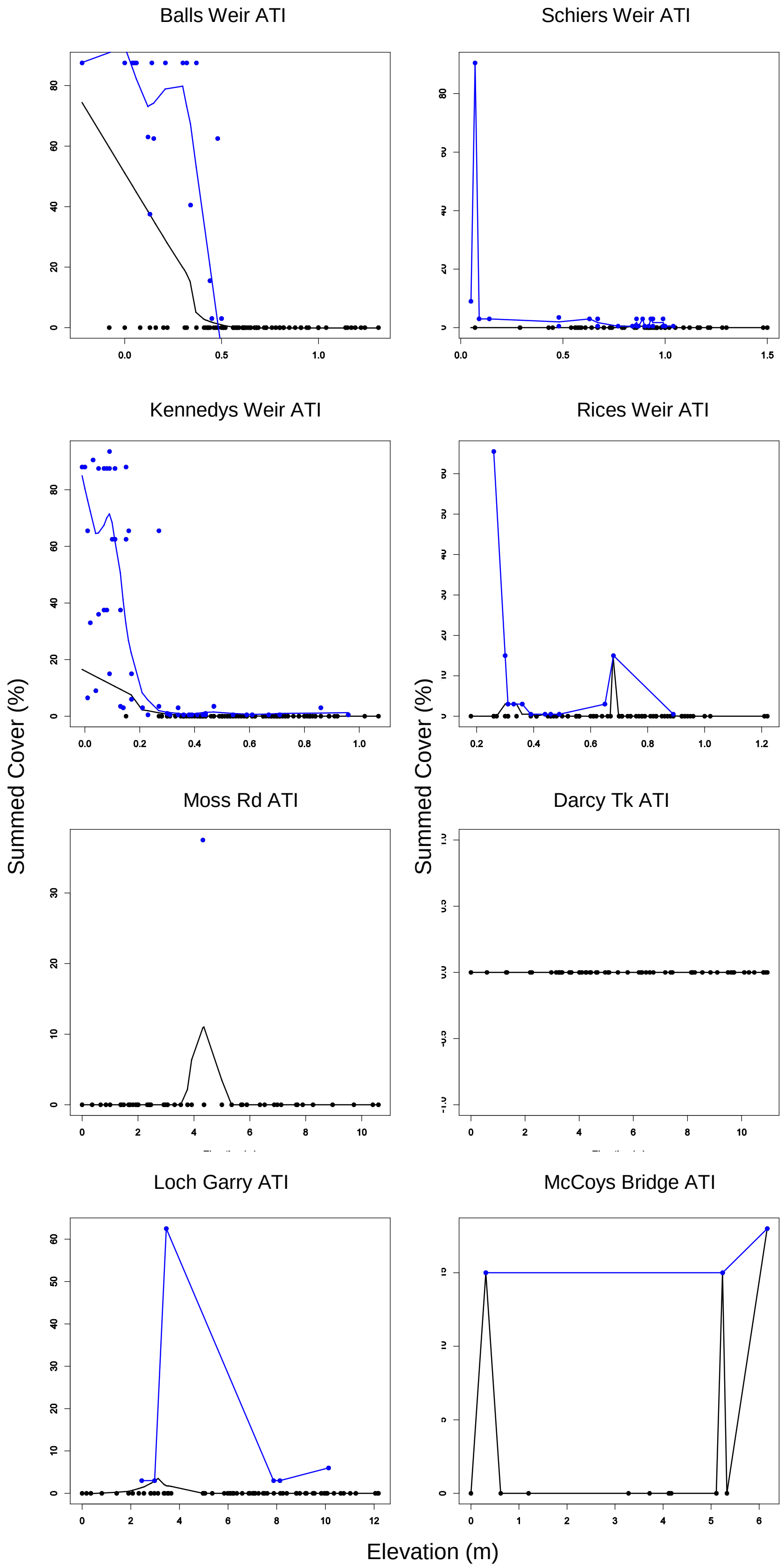

Figure C.4: Cover of Amphibious fluctuation tolerator, low growing species (y-axis) versus elevation from the water surface (x-axis) for the sites on Broken Creek (top four panels) and Goulburn River (bottom four). Points and lines are as described above.

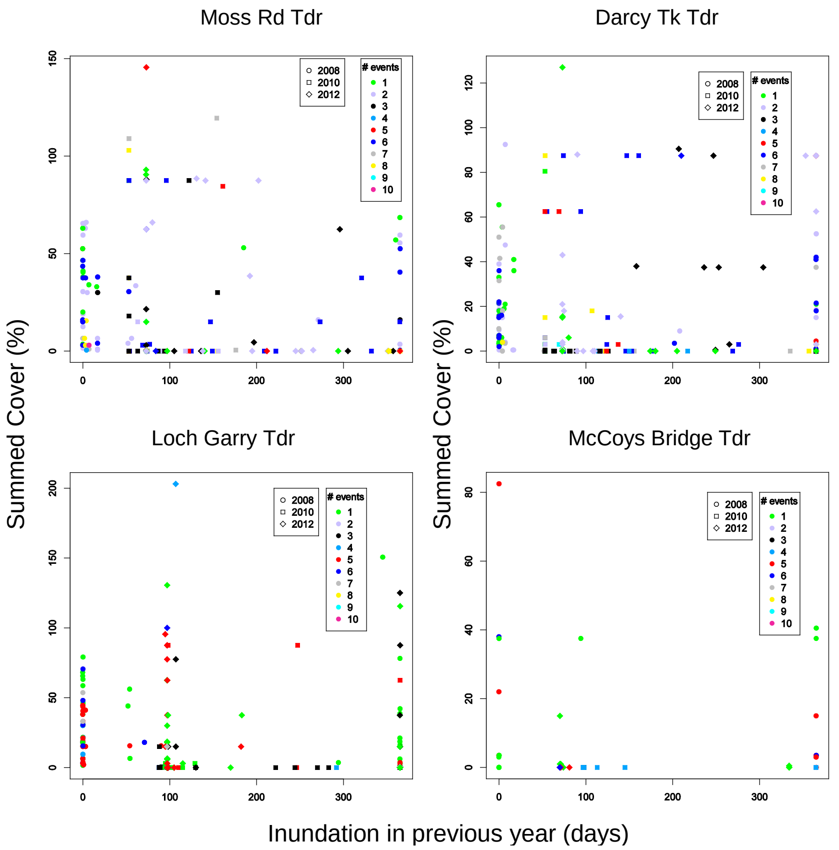

Figure C.5: Cover of Terrestrial dry species (y-axis) versus number of days of inundation per year (x-axis) for the sites on Goulburn River. Colour denotes number of inundation events. Shape denotes year of sampling.

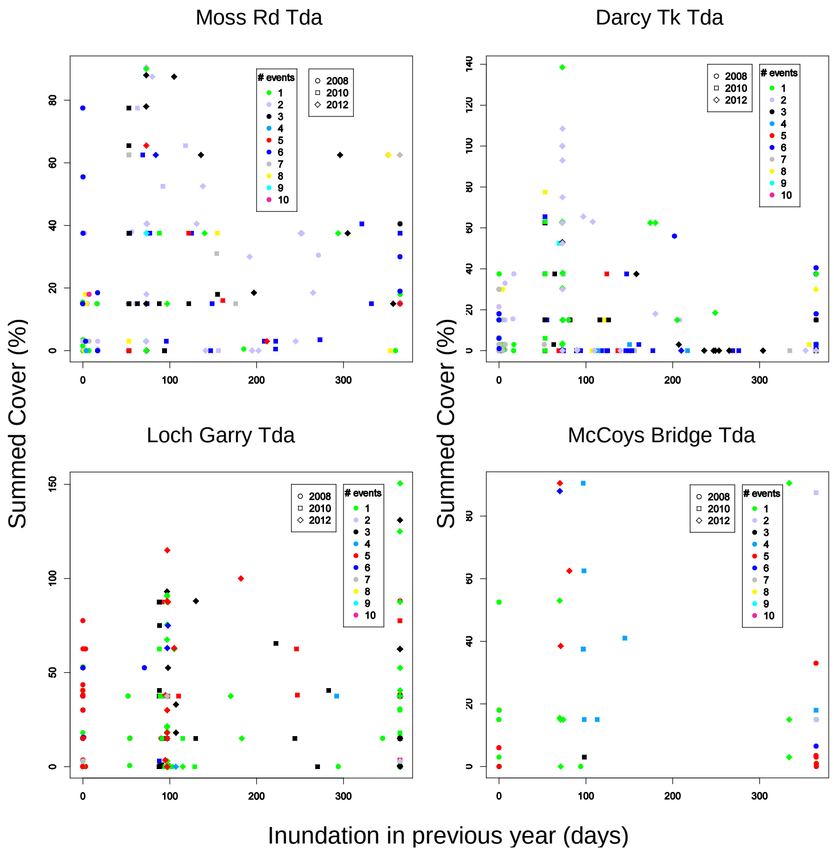

Figure C.6: Cover of Terrestrial damp species (y-axis) versus number of days of inundation per year (x-axis) for the sites on Goulburn River. Colour denotes number of inundation events. Shape denotes year of sampling.

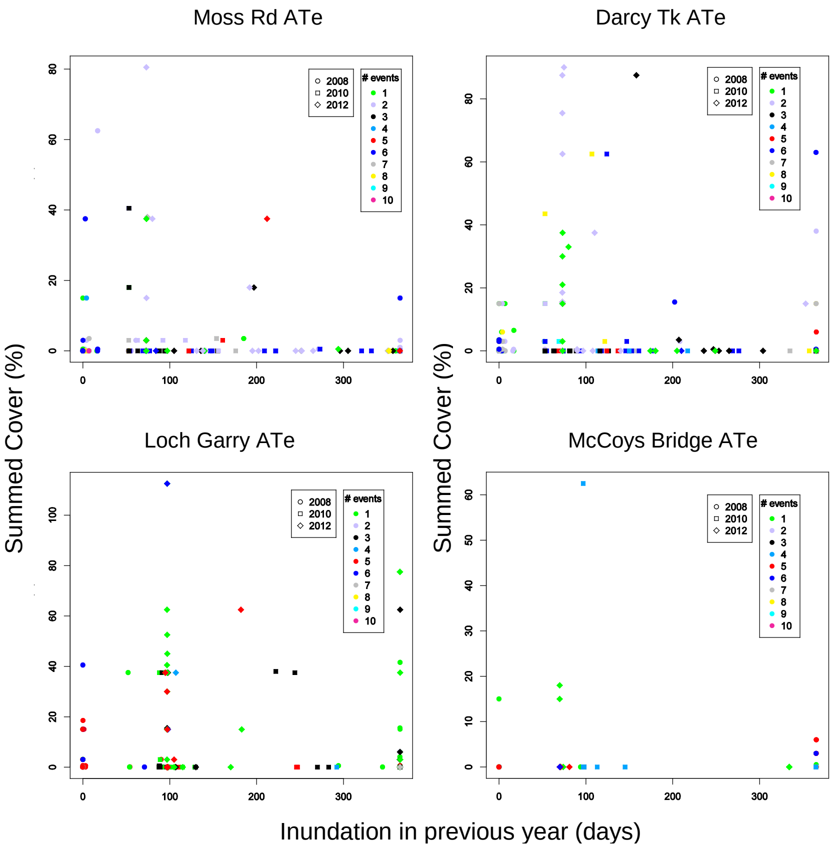

Figure C.7 Cover of Amphibious fluctuation tolerator, emergent species (y-axis) versus number of days of inundation per year (x-axis) for the sites on Goulburn River. Colour denotes number of inundation events. Shape denotes year of sampling.

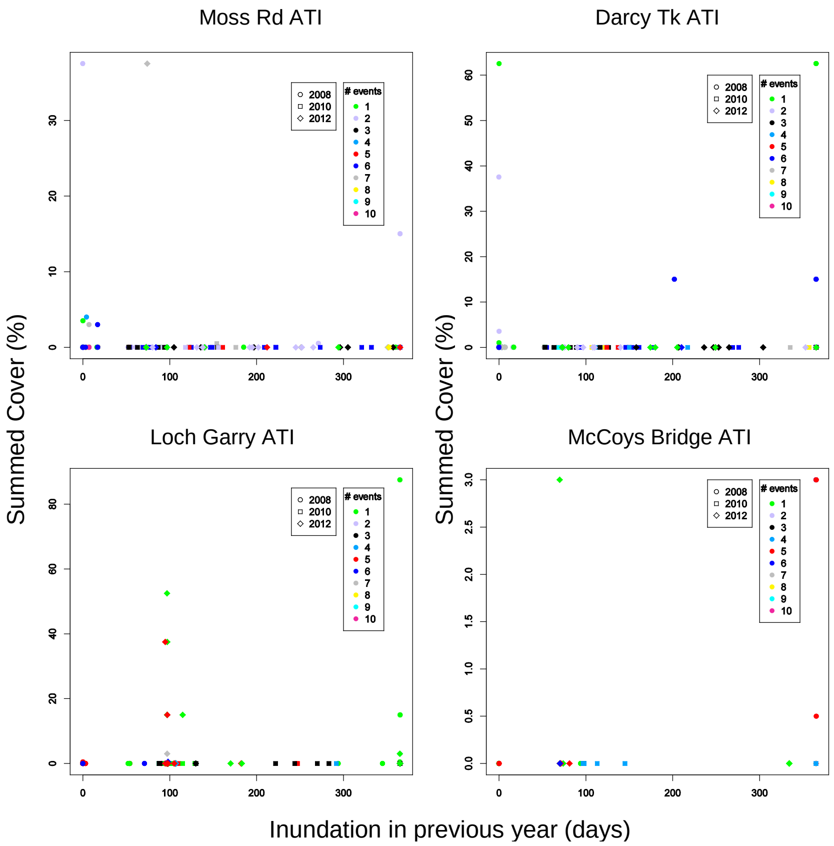

Figure C.8: Cover of Amphibious fluctuation tolerator, low-growing species (y-axis) versus number of days of inundation per year (x-axis) for the sites on Goulburn River. Colour denotes number of inundation events. Shape denotes year of sampling.