Institute for Land, Water and Society
Charles Sturt University

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# Monitoring of ecosystem responses to the delivery of environmental water in the Murrumbidgee system

Report 2 – May 2012

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# EXECUTIVE SUMMARY

In June 2011, nearly 110 gigalitres of Commonwealth environmental water were provided to a watering action managed by New South Wales, which totalled 161 gigalitres (including 23 gigalitres from the The Living Murray; 21 gigalitres from New South Wales Environmental Water Allowance and 8 gigalitres from private donations) targeting the mid-Murrumbidgee wetlands. The water was released from Burrinjuck and Blowering Dams with the environmental flow reaching a maximum daily discharge of 24,908 ML/day in the Murrumbidgee River downstream of Burrinjuck Dam on 17$^{th}$ June 2011 and 9,492 ML/day in the Tumut River downstream of Blowering Dam on 16$^{th}$ June 2011. From July 2011 to February 2012, natural and managed river flows further inundated a sub-set of wetlands in the mid-Murrumbidgee.

The monitoring program assessed two key components of the environmental release - the response of wetland flora, fauna and water quality after the water had filled key sites within the nationally important **mid-Murrumbidgee wetlands** (Environment Australia, 2001) and the response of biofilms and macroinvertebrates within the river channel (**in-stream**) as the water was released from Blowering and Burrunjuck Dams. These two components are covered separately within this report.

Summary of key outcomes from the June 2011 environmental release were:

- Improved water quality, including reduced dissolved organic carbon and stabilised dissolved oxygen levels.
- Increased native fish species diversity from 2005 survey levels (Gilligan 2005) and successful native fish recruitment.
- Successful recruitment for wetland frog species leading to an increased abundance of frogs within filled wetlands when compared to the control wetlands.
- Promoted nesting and created conditions that allowed for successful fledging of darters, great cormorants and little pied cormorants at key filled wetlands.
- Increased aquatic vegetation cover and species diversity within filled wetlands.
- Benefits to in-stream ecosystem due to reduced biomass of biofilms, increase in the relative proportion of early successional algal taxa (e.g. diatoms) and increased number of macroinvertebrate taxa.
- Scouring of biofilms also provided short-term benefit to the community by reducing the nuisance factor that occurs when the biofilms form mats and builds up to unacceptable levels.

### *Mid-Murrumbidgee wetlands*

Monitoring of the responses of wetland biota (vegetation, frogs, fish, waterbirds and freshwater turtles), organic carbon and water quality was undertaken at twelve wetlands in the mid-Murrumbidgee wetland complex between June 2011 and February 2012. Additional data was drawn from previous wetland monitoring surveys conducted between November 2010 and April 2011 by Wassens and Amos (2011). Three types of treatment were included in this study - **filled** (received environmental water during the release), **control 1** (did not receive environmental water during the release) and **control 2** (did not receive water during the environmental release, but received run-off from rainfall and drainage water from surrounding areas).

The key outcomes of the June 2011 environmental release combined with subsequent small scale natural river flow increases in late August/September and a small environmental release in December 2011 within the mid-Murrumbidgee wetlands are as follows:

- The June 2011 environmental release had a positive impact on **dissolved, total and particulate organic carbon** levels within wetlands. Very high levels of dissolved organic carbon can contribute to low dissolved oxygen events (commonly called black-water), the environmental flow had a dilution effect leading to an overall decrease in organic carbon levels within the filled wetlands and reducing the risks of future low dissolved oxygen events. In contrast dissolved organic carbon levels increased within the control wetlands which did not receive environmental water. **Dissolved oxygen** levels were similar between the filled and control wetlands and were within the normal range throughout the 2011-12 monitoring period.

- The June environmental releases assisted in the recovery of **aquatic vegetation communities**. That is, recovery of aquatic vegetation communities was greatest within the filled wetlands when compared to the control wetlands. In particular**, the percent cover of aquatic vegetation within filled wetlands increased significantly over time**, while the percent cover of aquatic vegetation within the control wetlands remained stable. The rate of recovery of aquatic vegetation was influenced by the length of time that the wetland had been dry prior to first re-filling in August 2010. Wetlands that had been dry for between three and five years had the greatest increase in aquatic vegetation cover after receiving the environmental flows, compared with those that had been dry for more than five years.

- Overall, the June 2011 environmental releases had a positive impact on **frog breeding and abundance**. Five frog species were recorded during the monitoring period. The abundance of barking marsh frogs and spotted marsh frogs increased significantly, compared to surveys in 2010-11 levels, in wetlands that received environmental water in 2011 but not in the control wetlands. Breeding activity in response to the environmental release commenced in August 2011 when ambient temperature started to increase, and tadpoles were recorded from October 2011 onwards. Further natural and managed river flows prompted calling by summer active species such as Peron's tree frogs. Tadpoles of the barking marsh frog, spotted marsh frog and Peron's tree frog were recorded in both filled and control 2 wetlands, but no tadpoles were recorded in the control 1 wetland.

- Five native and five introduced **fish species** were recorded during this study. The environmental release appeared to favour native over exotic species, and native fish were more abundant in filled wetlands than introduced species. Juveniles of all five native species were recorded in filled wetlands, with juveniles of carp gudgeon, bony bream and un-specked hardyhead making up 50% or more of the total catch for these species by February 2012.

- Two species of **freshwater turtle** were recorded (Macquarie River and long-necked turtles) within the filled wetlands. In addition, Macquarie River turtle hatchlings were detected in two of the filled wetlands, Yarrada and Molleys lagoons, during the December 2011 and February 2012 surveys.

- **Waterbird communities** were diverse in the mid-Murrumbidgee wetlands, with 36 species recorded during ground surveys undertaken from June 2011 – February 2012. **Colonial waterbird breeding** was recorded following the June 2011 environmental release with a small number of nests recorded in Gooragool, Yarrada and McKennas lagoons. Surveys to date have indicated that the mid-Murrumbidgee wetlands are in a recovery stage, with vegetation communities of the wetlands re-establishing following their filling in 2011. Most waterbirds prefer wetland habitats with aquatic vegetation, either as foraging habitat or to provide shelter from weather and predators. We expect the wetlands to become more attractive to waterbirds as wetland vegetation is further established.

These findings suggest that while the mid-Murrumbidgee wetlands are still in a recovery phase following an extended dry period, they still support high levels of diversity in terms of aquatic vegetation, frogs, native fish, waterbirds and freshwater turtles. The 2011 environmental releases had a positive impact on fauna, flora and water quality within the filled wetlands and created conditions suitable for frog, native fish, turtle and waterbird breeding. By creating conditions for successful recruitment, the environmental release is likely to have contributed to the longer-term recovery of the fauna and flora communities within the mid-Murrumbidgee wetland area. Small top-up flows in spring and summer extended wetland hydroperiod, in many cases increased recruitment outcomes for native fish, frogs and freshwater turtles, and helped to maintain waterbird diversity through summer months. Small spring and summer top-up flows were therefore crucial to the overall success of this watering event, and the provision of small top-up flows should be incorporated into future watering plans, especially in situations where the bulk of the environmental release occurs in autumn or winter.

Plate 1 Juvenile inland banjo frog from Molleys lagoon which received commonwealth environmental water between June 2011 and February 2012

***In-stream ecosystem***

The in-stream ecosystem responses to the environmental flow in June 2011 were as follows:

- The environmental flow had a positive effect by significantly reducing the biomass of **biofilm** at several sites, most likely due to scouring of biofilms from increased water velocity. Scouring of biofilm provides benefit by contributing nutrients and food into the water column, thus providing an important resource for downstream communities. It also provides benefit to the community by reducing the nuisance factor that occurs when the biofilms form mats and builds up to unacceptable levels. However, approximately five weeks after the recession of the environmental flow following a return to normal regulated dam operations, the biofilm biomass had increased to levels higher than observed prior to the environmental flow, particularly at the most upstream sites. In contrast, biofilm biomass remained consistently low throughout the study in the unregulated Goobarragandra River.

- The **biofilm** was comprised of 58 algal taxa including red algae, green algae, blue-green algae and diatoms. In the Murrumbidgee River, the environmental flow provided benefit by reducing the relative proportion of red, green and blue-green algae and increasing the proportion of diatoms, which are common in unregulated river systems. In the upstream reaches of the Tumut River, the environmental flow provided only short-term benefits because dam operations returned to normal regulated practices soon after the environmental flow. The environmental flow provided a longer-term benefit at Tumut sites 3, and 4, downstream of the confluence with the unregulated Goobarragandra River, because the unregulated flows from the Goobarragandra River contributes to the health of these downstream reaches.

- The environmental flow had a short-term benefit on the in-stream ecosystem by increasing in the number of macroinvertebrate taxa in the Tumut River immediately after the environmental flow. However, the total number of taxa reduced after dam operations returned to normal regulated practices. For the majority of the dominant **macroinvertebrate** taxa there was also a short term increase in abundances immediately after the environmental flow.

These findings suggests that while **the environmental flow provided a short-term benefit to the in-stream ecosystem** through scouring of biofilms, the benefits were not sustained when the dam operations returned to normal regulated practices. The in-stream benefits of the environmental release were far greater in the Murrumbidgee River than in the Tumut River. This occurred because the environmental release from Blowering Dam to the Tumut River had a similar river height and discharge to normal delivery for consumptive use. This highlights the importance of understanding the normal operational flow regimes that are present within a regulated river system and understanding of the key flow components that reflect the variability of a more natural flow hydrograph. Environmental releases should specifically aim to restore key flow components that have been lost through river regulation and avoid delivering flows that would exacerbate negative impacts of regulated operations. By using a careful release strategy, environmental flows can be delivered to optimise benefits for both wetland and in-stream ecosystems.

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# List of Abbreviations

| | |
|---|---|
| AFDW | Ash Free Dry Weight |
| ANOSIM | Analysis of Similarity |
| ANOVA | Analysis of Variance |
| ANZECC | Australian and New Zealand guidelines for fresh and marine water quality |
| CEWH | Commonwealth Environmental Water Holder |
| CRSA | Colonisable Rock Surface Area |
| CSU | Charles Sturt University |
| DO | Dissolved Oxygen |
| DOC | Dissolved Organic Carbon |
| DW | Dry Weight |
| EPBC | Commonwealth *Environment Protection and Biodiversity Conservation Act 1999* |
| IMEF | Integrated Monitoring of Environmental Flows |
| ML | MegaLitre |
| NOW | NSW Office of Water |
| NOx | Nitrite |
| NTU | Nephelometric Turbidity Unit |
| OEH | NSW Office of Environment and Heritage |
| POC | Particulate Organic Carbon |
| SE | Standard Error |
| SF | State Forest |
| SIMPER | Similarity Percentages |
| TOC | Total Organic Carbon |
| TSC | NSW *Threatened Species Conservation Act 1995* |

# 1. Introduction

This monitoring program considers the physicochemical and biotic outcomes of the Commonwealth environmental water release within the Murrumbidgee catchment in June 2011. Two key aspects of release are considered: (1) the response of wetland biota, water quality, carbon and nutrients within wetlands that filled as a result of the environmental release in the mid-Murrumbidgee (between Wagga Wagga and Carathool), and (2) the response of in-stream biota downstream of Blowering and Burrunjuck dams, which were used to supply water for the release. The locations of the sites used in each aspect of the study are illustrated in Figure 1. These two components of the ecological responses to environmental releases are considered separately throughout this report.

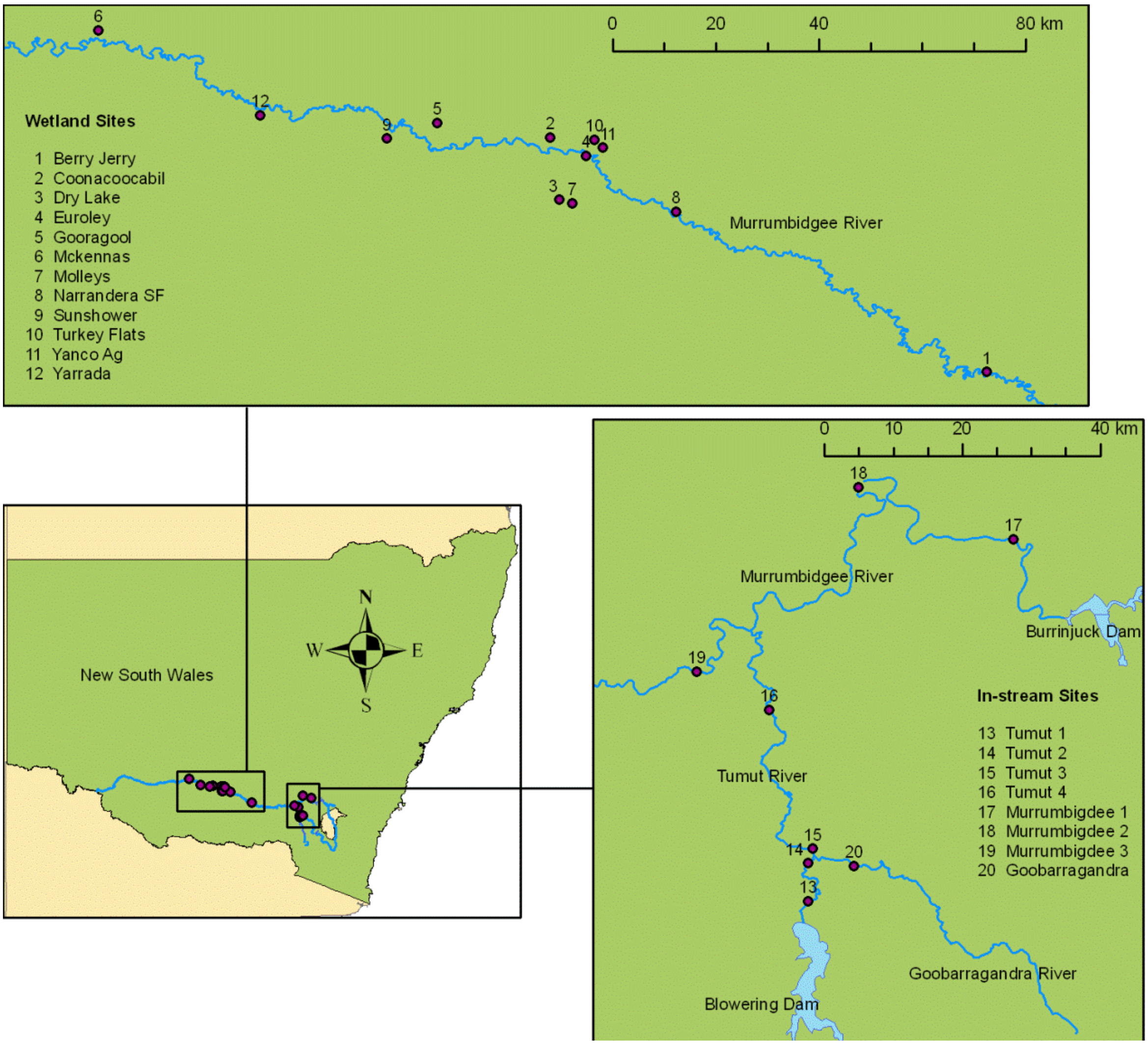



Figure 1. Location map of in-stream and wetland sites.

## 1.1. Wetland flora and fauna responses to flooding

Water regimes within the mid-Murrumbidgee River have been greatly altered by river regulation and water extraction. In particular, there have been changes to the timing of flows and the frequency of wetland filling events. For example, at Gundagai, summer and autumn flows have increased and flows during winter and spring have been reduced (Frazier and Page 2006; Frazier, Page *et al.* 2005; Page, Read *et al.* 2005). There has been a significant reduction in the frequency of intermediate and large flood events (Page, Read *et al.* 2005). Within the river channel these changes in water regime have led to changes in water velocities and the availability of particular habitat types. The regulation of flows has also reduced the frequency and duration of inundation of the mid-Murrumbidgee wetlands. Some wetlands have experienced a reduction in flooding due to the altered flow regime, and others have had an increase in hydroperiod due to their use as water storages or as a result of receiving draining water at the end of irrigation seasons. For wetlands between Gundagai and Hay, with river connections higher than the level of irrigation flows, there has been a halving of the average frequency of inundation (Frazier and Page 2006). Thornton and Briggs (1994) noted that 62% of wetland area in the mid-Murrumbidgee has been subject to some level of hydrological modification; in the early 1990s, 34% of this area had been made more permanent.

Significant decreases in the frequency of intermediate flood events had been reported for the period between 1970 and 1998 (Page, Read *et al.* 2005). However, severe drought conditions occurred throughout the region between 2000 and 2010. Although a small flow peak reached low lying wetlands in September 2005, the majority of wetlands did not refill until significant natural flood events occurred between August 2010 and January 2011 (Figure 2). The response of wetland biota and changes in water quality and carbon levels must therefore be considered in the context of wetlands that have undergone significant extended drying that far exceeded their natural flood return interval. Within this context the responses of wetland biota and changes in water quality and carbon levels need to be considered as part of the long-term recovery of these wetlands, rather than the final outcomes of a specific environmental release.

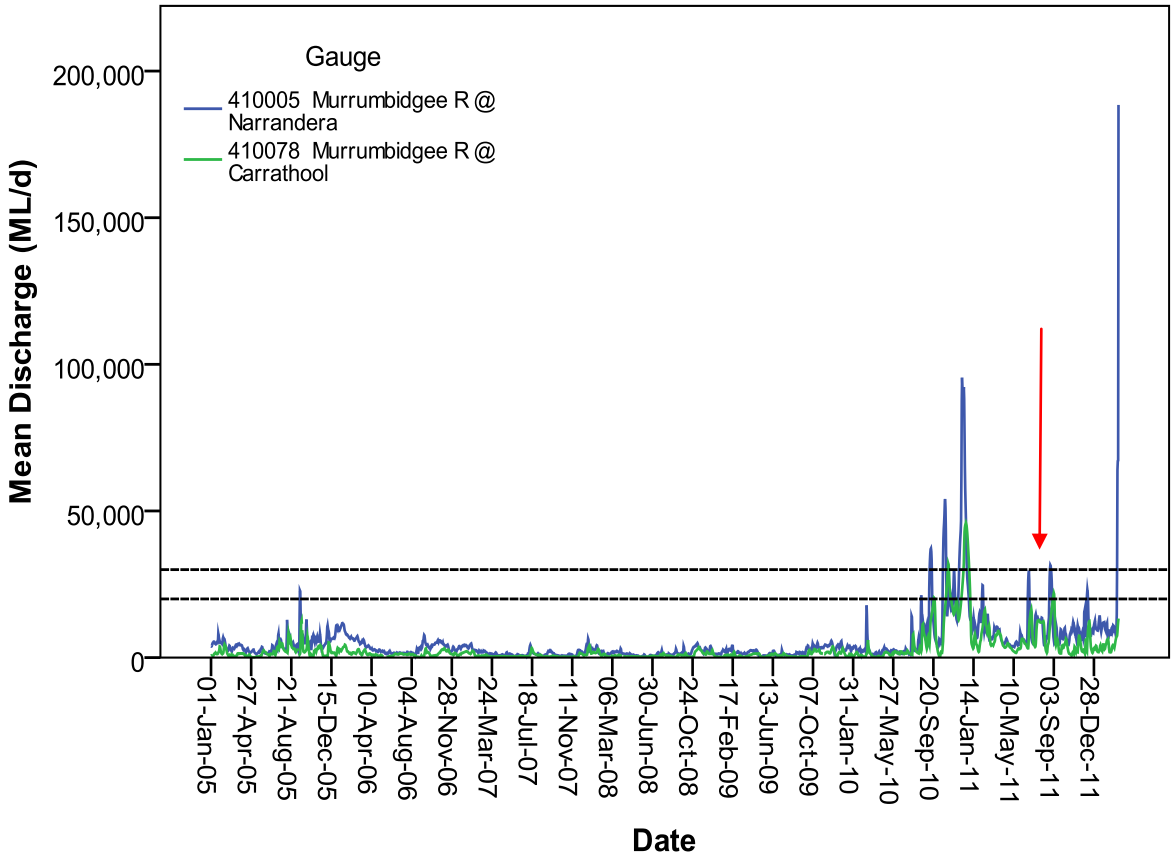



Figure 2. Murrumbigee River discharge at Narrandera and Carrathool between 01/01/2005 and 09/03/2012. Dashed lines show commence to fill levels for wetlands within the Mid-Murrumbidgee. Lower line is commence-to-fill for the lowest lying wetlands, upper line is the commence-to-fill for the higher wetlands from (Murray 2008). The red arrow shows the environmental water release in June 2011.

While wetland flora and fauna are generally considered to be well adapted to highly variable wetting and drying cycles, individual species differ in their thresholds to drying and their ability to recover following rewetting. There are two key processes that influence fauna and flora responses to flooding: (1) the level of connectivity by flood water that facilitates dispersal and re-colonisation of wetlands by aquatic species (Sheldon, Boulton *et al.* 2002); and (2) the in-situ conditions within a particular wetland before, during and after flooding that can influence seed and egg bank persistence, viability and germination (Brock, Nielsen *et al.* 2003).

Refugia are habitats that support a population during periods of disturbance, such as drought, for fully and semi-aquatic species such as fish, frogs and turtles. During dry periods refugia can take the form of a permanent waterbody (Sheldon, Bunn *et al.* 2010), or, in some instances, artificial habitat such as farm dams and irrigation infrastructure (Wassens 2006). During dry periods, aquatic individuals are effectively confined to their refuge habitats, which are typically disconnected from other water bodies. Following flooding, dispersal is the key mechanism that allows these species to move out of refugia and into the newly-flooded habitats. For example, dispersal of Southern bell frogs *Litoria raniformis* is closely linked to flooding, with individuals moving large distances to recolonise wetlands following a flood event (Wassens, Watts *et al.* 2008). As a consequence, the recovery of aquatic and semi-aquatic fauna may be closely linked to the quality of the refuge habitat and the level of connectivity between it and surrounding wetlands. Small flow pulses will connect fewer wetlands than large events. Hence, the flow pulse size will influence the level of re-colonisation of wetlands and the recovery of wetland populations (Magoulick and Kobza 2003).

The production of resistant propagules (seeds, eggs and spores) that can survive in the soil between floods and during drought is common for many aquatic plants, zooplankton, phytoplankton and some macro-invertebrates (Brock, Nielsen *et al.* 2003). Individual species have specific cues that are required to break their dormancy, and, as a result, not all species within the seed or egg bank will geminate or hatch at the same time. Instead, multiple successive communities may establish with successive filling events (Brock, Nielsen *et al.* 2003). The viability of the seedbank can also be influenced by disturbance (for example, grazing and cultivation) (Tuckett, Merritt *et al.* 2010) and the hydrological characteristics of the filling event itself (for example, depth, duration and oxygen) (Casanova and Brock 2000). As a result, the community that emerges from the seed and egg bank is unlikely to contain all the species present. The viability of seeds and eggs declines over time (Casanova and Brock 2000). Some studies have demonstrated that up to 50% of the seed bank can be lost after ten years of drought (Leck and Brock 2000). Repeat flooding over a number of years may be required to fully restore wetland plant communities and replenish depleted seed and egg banks following extended droughts (Alexander, Nielsen *et al.* 2008).

As the wetlands included in this study were dry prior to 2010, the composition of fish communities within the wetlands will be strongly influenced by connectivity between each wetland and the Murrumbidgee River and drought refuges, and the maintenance of viable populations within these permanent habitats between flood events. Re-colonisation of wetlands by fish is also influenced by the timing of flooding, with movements into wetlands typically lower in winter than in spring and summer (Humphries, King *et al.* 1999). Once fish enter the wetland or anabranch, breeding success and survival can be influenced by water quality, particularly dissolved oxygen and water temperature, depth, food availability and competition and predation (King, Humphries *et al.* 2003).

Many native fish species prefer to breed as water recedes rather than immediately after a flood pulse, and, as a consequence, recruits are often more abundant in late summer (Humphries, Serafinia *et al.* 2002).

Fish communities within the Murrumbidgee catchment were last assessed in 2004 (Gilligan 2005). These surveys described fish communities in the mid-Murrumbidgee as being severely degraded with eight of the 21 native fish species previously recorded being either locally extinct or occurring in very small numbers. Only five fish species were recorded from three wetlands and these were dominated by introduced fish species, mainly Gambusia (*Gambusia holbrooki*), along with European carp (*Cyprinus carpio*) and goldfish (*Carassius auratus*). One native genera, carp gudgeon, (*Hypseleotris* spp.), and one native species, Australian smelt (*Retropinnia semoni*) were also recorded (Gilligan 2005). The proportion of native species increased downstream and were higher in wetlands across the lower Murrumbidgee floodplain, which have retained a somewhat more natural flooding regime than those in the mid-Murrumbidgee (Gilligan 2005). Spencer and Wassens (unpublished data, 2011) recorded six native fish species and one native genera: Bony bream (*Nematalosa erebi*), Australian smelt, Flat-head gudgeon (*Philypnodon grandiceps*), Murray-Darling rainbow fish *Melanotaenia fluviatilis*, Unspecked hardyhead (*Craterocephalus stercusmuscarum fulvas*), Golden perch (*Macquaria ambigua ambigua*) and Carp gudgeons in newly-flooded wetlands in the lower Murrumbidgee floodplain (downstream of Hay) between 2008 and 2011 (Spencer and Wassens, 2010).

Like fish, frog populations within wetlands can be influenced by both connectivity, which influences the ability of individuals to move into newly-flooded habitats (Smith and Green 2005; Smith and Green 2006; Wassens, Watts *et al.* 2008), and the conditions within the wetland (Jansen and Healey 2003; Wassens, Hall *et al.* 2010; Wassens and Maher 2011). Aquatic vegetation complexity (Jansen and Healey 2003), abundance of introduced fish, such as Gambusia (Reynolds 2009) and European carp (Spencer and Wassens, 2010) can all influence the occupancy of wetlands and the level of recruitment (tadpole abundance and survival through to metamorphosis). Frogs have defined activity periods, and it is common to see a succession of species breeding within a single wetland between autumn and summer (Paton and Crouch 2002).

The mid and lower Murrumbidgee floodplain supports upwards of ten frog species, including the nationally vulnerable Southern bell frog (Commonwealth Environment Protection and Biodiversity

Conservation (EPBC) Act 1999). Frog communities were assessed as part of the Integrated Monitoring of Environmental Flows (IMEF) program by James Maguire (NSW Office of Environment and Heritage (OEH)) and Lorraine Hardwick (NSW Office of Water, (NOW)) between 1998 and 2005, and eight frog species were recorded in wetlands of the mid-Murrumbidgee. Jansen and Healy (2003) recorded six frog species in their assessment of wetlands along the Murrumbidgee River, and identified the complexity of aquatic vegetation and grazing intensity as key predictors of frog species occurrence within a wetland and subsequent breeding success.

Organic matter is a critical water quality parameter for understanding the functioning of river and wetland ecosystems. The dominant form of organic matter in many aquatic ecosystems is dissolved organic carbon (DOC) and this complex mixture of organic compounds forms an important source of energy for microbial processes (Findlay and Sinsabaugh 1999). The mixture of compounds varies between sites, but consists of between 50 – 90% humic and fulvic substances (large, poorly defined substances) with the remainder consisting of simpler compounds such as amino acids, fatty acids, hydrocarbons and carbohydrates (Morel and Hering 1993). The humic and fulvic substances are less readily metabolised than the simpler organic material (Morel and Hering 1993) and so a study of the humic and fulvic character of organic matter can provide clues to the bioavailabilty of the organic matter within an aquatic system. These substances can be formed as material is processed and aged within the ecosystem, but is also prone to degradation in sunlight with the production of more bioavailable materials (Howitt, Baldwin *et al.* 2008). Dissolved organic matter serves many functions within aquatic ecosystem including the protection of aquatic organisms from damage by ultraviolet light (Laurion, Ventura *et al.* 2000), and influencing the transport, degradation and toxicity of organic pollutants and dissolved metals (Choudhry 1984). In addition, many Australian riverine ecosystems are reliant on both algal and dissolved organic matter for microbial productivity and can even be limited by very low DOC concentrations (Hadwen, Fellows *et al.* 2010).

Dissolved organic matter has a positive effect on ecosystem health by promoting high primary productivity. However, large amounts of organic carbon can accumulate in floodplain wetlands as litter and coarse woody debris, with floods releasing substantial amounts of dissolved organic carbon (Robertson, Bunn *et al.* 1999). Under certain conditions very high organic matter inputs coupled with high water temperatures can lead to a rapid increase in microbial metabolism leading to decreases in dissolved oxygen concentration (often referred to as blackwater events) (Hladyz, Watkins *et al.* 2011; Howitt, Baldwin *et al.* 2007). In some circumstances this effect may be severe enough to result in negative impacts on aquatic organisms and even fish kills (Baldwin, Howitt *et al.* 2001). These conditions may be especially prevalent in areas where wetlands and their floodplain have been dry for extended periods, allowing large amounts of organic matter to accumulate, which are then subject to flooding during the summer months when water temperatures are high. In this context the application of environmental water during the cooler months can assist in lowering dissolved organic carbon levels to a point where they will have a beneficial impact on aquatic ecosystem function while reducing the risks of low dissolved oxygen events as temperatures increase.

## 1.2. In-stream responses to flows

In-stream river ecosystems downstream of dams generally have reduced aquatic macro-invertebrate diversity (Boon 1988; Doeg 1984), high biomass of nuisance biofilms, and reduced diversity of biofilms (Ryder 2004; Watts, Ryder *et al.* 2008).

Biofilms (a combination of bacteria, algae, fungi and detritus growing on submerged surfaces) are excellent indicators of flow management because they respond to flow changes in a short time frame (days to weeks) that is appropriate for the management change. They are central to important nutrient and biogeochemical processes and are a major food resource for higher organisms including crustaceans, insects and some fish. There are several reasons why it is desirable to scour biofilms from cobble substrate through the release of environmental flows from dams:

- To promote early successional algal taxa (e.g. diatoms) and higher biofilm diversity. A high diversity of biofilms usually indicates good ecosystem health.

- To contribute nutrients and food into the water column, thus providing an important food resource for downstream communities

- To reduce the nuisance factor that occurs when biofilm growing on the beds of rivers builds to levels that are unacceptable to the general public or landholders, Quinn (1991) recommended that "the seasonal maximum cover of stream or river bed by periphyton as filamentous growths or mats (greater than about 3 mm thick) should not exceed 40% and/or biomass should not exceed 100 mg chlorophyll-*a* /m$^2$ .

Freshwater macroinvertebrates are good indicators of river health and flow management because: they are an important component of riverine foodwebs; they are abundant and diverse; their taxonomy is well known; and many taxa are sensitive to stress and respond to changes in environmental conditions.

Disturbance by flood events is one of the most important regulators of spatial and temporal variability in benthic communities of streams (Davis and Barmuta 1989), with shifts in benthic algal community structure and function being well documented e.g. (Biggs, Smith *et al.* 1999; Peterson and Stevenson 1992; Uehlinger, BÜhrer *et al.* 1996). There have been many investigations of the effects of water velocity on biofilm biomass and productivity in flumes e.g. (Horner, Welch *et al.* 1990)), experimental streams (e.g. Biggs, Smith *et al*. 1999) and by survey in natural streams e.g. (Uehlinger, Kawecka *et al.* 2003). The experimental flood downstream of the USA's Glen Canyon Dam in the mid-1990s reported some of the first examples of the potential for biofilm scour, with benthic scour and entrainment of both primary and secondary consumers occurring along the 385-kilometre river corridor impacted by the flood (Uehlinger, Kawecka *et al.* 2003).

In Australia, research by (Sutherland, Ryder *et al.* 2002) and (Watts, Nye *et al.* 2005; Watts, Ryder *et al.* 2009; Watts, Ryder *et al.* 2008) provide the most comprehensive series of investigations on the effects of pulsed flow releases on biofilms. Increasing the variability and magnitude of flows through the release of pulsed flow events was shown to scour and reset biofilms. This had a positive effect on the instream ecosystem by reducing the biomass of biofilm and enabling early successional algae (e.g. diatoms) to become established, thereby facilitating a shift in the biofilm community towards that of a reference stream (Watts, Nye *et al.* 2005; Watts, Ryder *et al.* 2008; Watts, Zander *et al.* 2011). Pulsed flows produce velocities sufficient to scour biofilm biomass from the cobble substrata and change the community composition of biofilms by removing filamentous green and blue-green algae and increasing the relative biovolume of early successional species of diatoms. However, the changes to biofilms are often not sustained, with the biofilms returning to pre-flow levels 30- 37 days after the flow peak (Sutherland, Ryder *et al.* 2002; Watts, Nye *et al.* 2005).

One of the most comprehensively documented studies of invertebrate responses to experimental pulsed flows was undertaken on the Spöl River in Switzerland. Overall, the Spöl River studies concluded that experimental flooding was beneficial to the aquatic invertebrate community (Jakob, Robinson *et al.* 2003; Robinson, Aebischer *et al.* 2004). Although reductions in invertebrate abundance immediately after flow pulses has often been observed e.g. (Matthaei, Uehlinger *et al.* 1997), the invertebrate community usually recovers to pre-flood levels within two months.

In Australia, Sutherland *et al.* (2002) reported shifts in the invertebrate community towards reference condition following a managed flow pulse in the Mitta Mitta River. (Chester and Norris 2006) measured the response of macroinvertebrates to monthly flow spikes released from one of the dams in the Cotter River. Sites downstream of the dam that were exposed to the flow pulse had more macro-invertebrate taxa and less periphyton chlorophyll-a content than sites downstream of dams without managed environmental flows, suggesting that a more suitable food supply resulting from environmental flow releases shifted macroinvertebrate communities towards those of unregulated streams.

Managed flow pulses have also been undertaken in the Cudgegong River, NSW, using the pulsed release of irrigation flows from Windamere to Burrendong Reservoirs to ameliorate the impacts of river regulation. The pulsed flows altered the abundance and richness of the invertebrate community, but there was variability in the responses of different taxa and across different habitats and years (Boey and Chessman 1997; Moroney, Royal *et al.* 1995). For example, Moroney *et al.* (1995) noted that mayfly abundance increased in response to a pulsed flow in 1994/1995, but Boey and Chessman (1997) recorded a decrease in mayflies in 1996. Moroney *et al.* (1995) found that the abundance of all invertebrate groups recovered after flooding, except for the stoneflies and molluscs. Both Moroney et al. (1995) and Boey and Chessman (1997) showed that dipteran abundance increased in response to pulsed flows. Overall, the Cudgegong studies concluded that the pulsed flows had no detrimental impact on invertebrate diversity (Boey and Chessman 1997; Moroney, Royal *et al.* 1995; Norris and Thoms 2003).

# 2. METHODS

## 2.1. Monitoring of the mid-Murrumbidgee Wetlands

The mid-Murrumbidgee wetlands component of this study monitored water quality and carbon, vegetation, frogs, fish, waterbirds and freshwater turtles to assess the ecological response of wetlands to the environmental flow event in the Murrumbidgee River in June 2011 and subsequent managed and natural river flows through to December 2011. The following hypotheses were examined in this study:

- Aquatic and semi-aquatic vegetation cover and species diversity will increase in spring and summer following the environmental release when compared with 2010-11 data.
- Vegetation communities will differ between wetlands filled during the environmental release and the controls (wetlands that did not fill during the release).
- Frog communities will respond to the environmental release in spring and summer and commence breeding activity (as measured by the presence of tadpoles and calling activity) within filled wetlands.
- Recruitment (as measured by the abundance of juveniles) for native and exotic fish species will occur within wetlands filled via the environmental release.
- Waterbird diversity will increase within filled wetlands and breeding will occur where water remains pooled through spring and summer.
- Dissolved organic carbon, total organic carbon and particulate organic carbon levels will decrease over time within wetlands filled as a result of the environmental release.

### 2.1.1. Sites and hydrology

This study is focused on twelve wetlands that make up part of the mid-Murrumbidgee wetland complex between Wagga Wagga and Carathool (see Figure 3). The majority of surrounding land is cleared grazing and cropping (see Figure 3 insets). Wetlands on both private land and within the NSW national river red gum reserve system are subject to periodic cattle grazing.

The hydrology of some wetlands included in this study is influenced by water extraction and storage. Nine of the twelve wetlands are moderately deep ox-bow lagoons, two are large open depressions (Turkey Flat and Dry Lake) and two are prior stream channels (Molleys) and Coonancoocabil (Plate 2). River red gum (*Eucalyptus camaldulensis*) is the dominant vegetation community surrounding these wetlands.

The release of environmental water from Burrinjuck and Blowering Dams commenced on June 14$^{th}$ 2011 (see figure 4). The environmental flow reached a maximum daily discharge of 24,908 ML/day in the Murrumbidgee River downstream of Burrinjuck Dam on 17$^{th}$ June 2011 and 9,492 ML/day in the Tumut River downstream of Blowering Dam on 16$^{th}$ June 2011. Natural and managed river flows contributed to top-ups of wetlands in late August 2011 and December 2011.

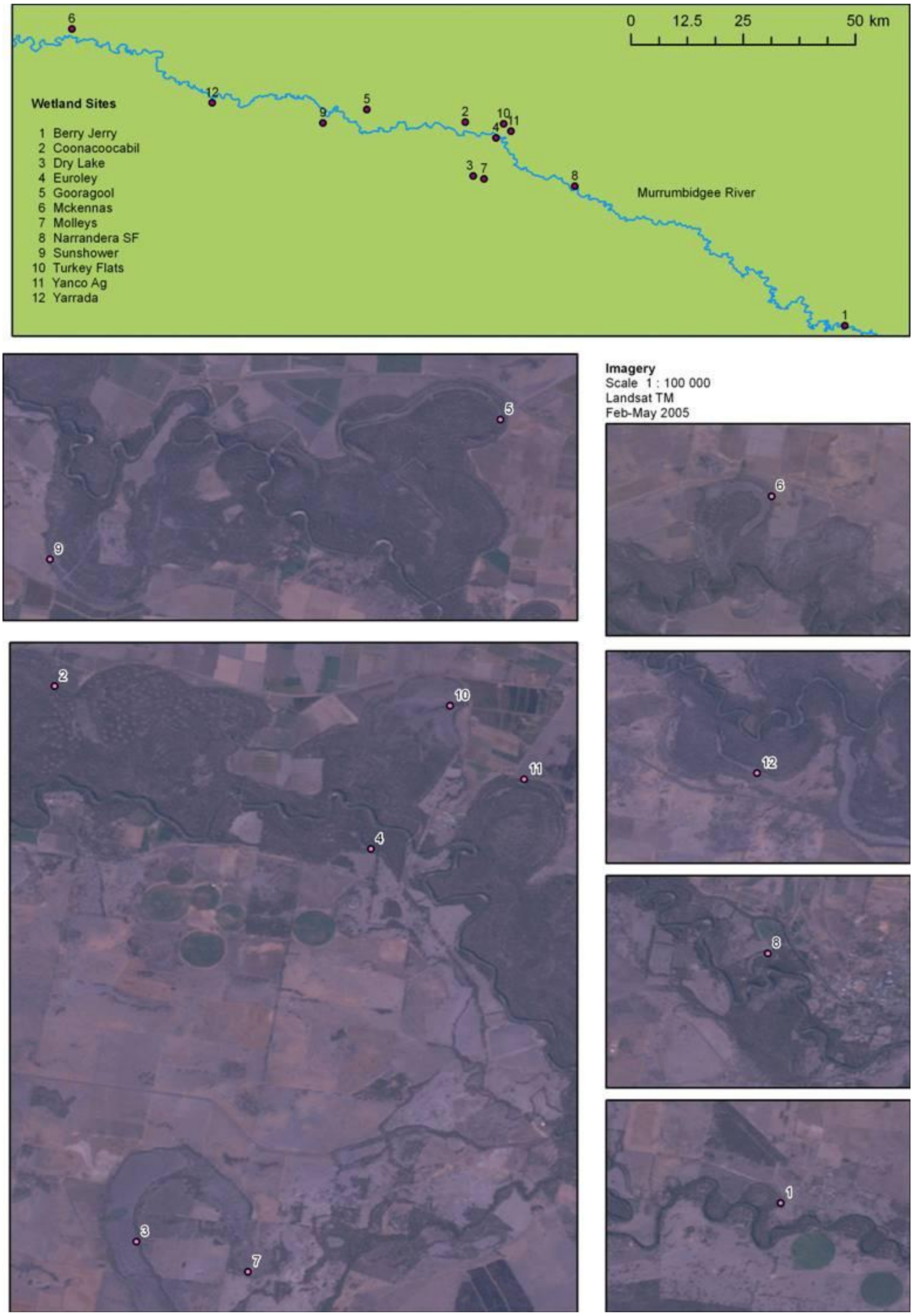



Figure 3. Distribution of wetland monitoring sites along the Murrumbidgee River. LandSat derived imagery provides individual site context.

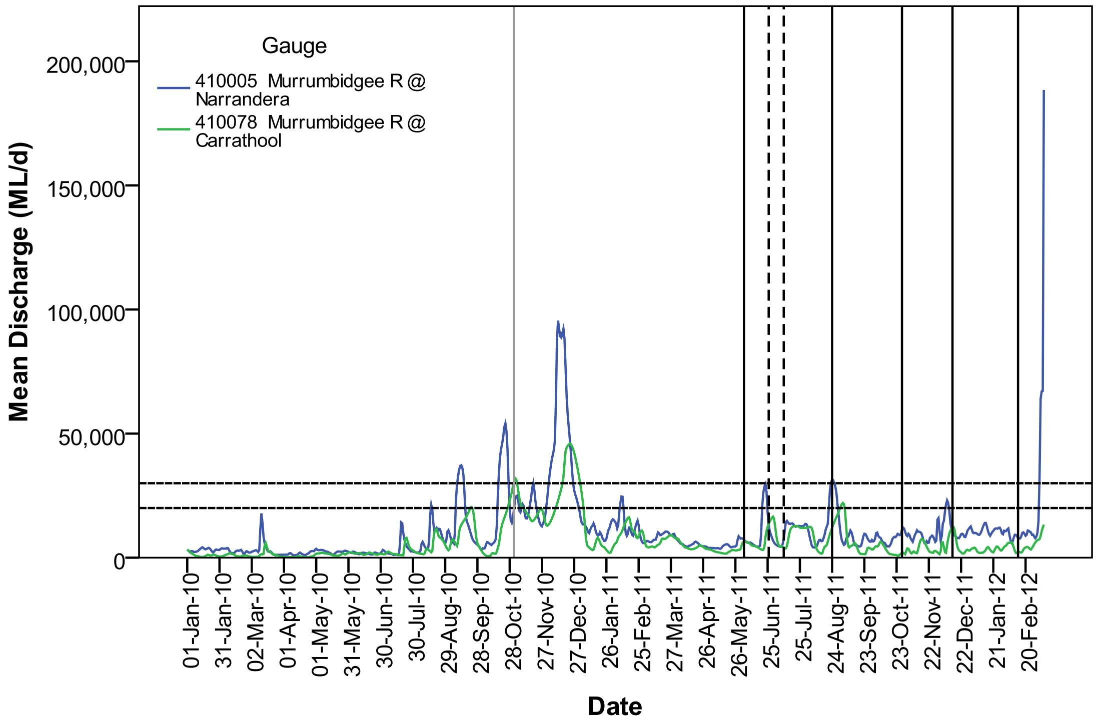



Figure 4. Timing of survey events in relation to discharge in the Murrumbidgee River at Narrandera and Carrathool. Grey vertical line shows commencement of monitoring by Wassens and Amos (2011), black vertical lines show monitoring as part of this project – solid lines (water quality and wetland biota), dashed vertical lines (water quality and carbon only). Dashed horizontal lines show commence to fill levels for wetlands within the mid-Murrumbidgee: lower line is commence-to-fill for the lowest lying wetlands, upper line is the commence-to-fill for the higher wetlands from Murray (2008).

Nine of the twelve wetlands monitored during this study were inundated by the environmental flow release in June 2011 (Plate 2, (a)-(g)). Due to natural and managed river flows, the same nine wetlands received further water in late August/September 2011 and five wetlands received water in December 2011 (Figure 4, Table 1). Two types of control wetlands have been included in this study: (1) Control 1 (Euroley, Plate 2(h)) was inundated during the natural flood event in December 2010 but was not inundated by the environmental flow in June 2011; and (2) Control 2 wetlands (Turkey Flats and Yanco Ag, Plate 2, (i)-(j)) were not inundated via the Murrumbidgee River in December 2010 or the 2011 environmental releases but are subject to relatively frequent low level inundation via rainfall run-off and as managed flows through the Murrumbidgee Irrigation Area.

Table 1. Summary of survey sites and number of surveys completed June 2011 - February 2012.

| | Site name | Top-up December | Humic and fulvic character | Carbon and Nutrients | Waterbirds | Frogs & tadpoles (broad-scale) | Fish | Aquatic vegetation |
|---|---|---|---|---|---|---|---|---|
| **Flooded Wetland (mid-Murrumbidgee)** | Berry Jerry | Yes | 7 | 4 | 5 | 5 | - | 4 |
| | Dry Lake | **No** | 6 | 4 | 5 | 5 | 2[a] | 4 |
| | Gooragool | Yes | 7 | 4 | 6 | 5 | 3[b] | 4 |
| | Mckennas | **No** | 7 | 4 | 6 | 5 | 3[c] | 4 |
| | Molleys | Yes | 7 | 4 | 6 | 5 | 4 | 4 |
| | Narrandera SF | **No** | 7 | 4 | 5 | 5 | - | 4 |
| | Sunshower | Yes | 7 | 4 | 6 | 5 | 4 | 4 |
| | Coonacoocabil | **No** | 5 | 3 | 5 | 5 | - | 4 |
| | Yarrada | Yes | 7 | 4 | 5 | 5 | 4 | 4 |
| **Control 1** | Euroley | **No** | 6 | 4 | 6 | 5 | 2[a] | 4 |
| **Control 2** | Turkey Flats | **No** | 6 | 4 | 6 | 5 | - | 4 |
| | Yanco Ag | **No** | 5 | 4 | 6 | 5 | - | 4 |

[a] Too shallow to sample December & February

[b] Too shallow to sample October

[c] Too shallow to sample February


(a) Berry Jerry


(b) Narrandera State Forest


(c) Molleys


(d) Dry Lake


(e) Coonancoocabil (June 2011)


(f) Sunshower

Plate 2 Survey wetlands in the mid-Murrumbidgee (August 2011) (a-f).

(g) Gooragool

(h) Yarrada

(i) Mckennas

(j) Euroley (Control 1)

(k) Turkey Flat (Control 2)

(l) Yanco Ag (Control 2)

Plate 2 (cont.) Survey wetlands in the mid-Murrumbidgee (August 2011) (g-l)

The methodology that follows was used previously as part of the IMEF (1998-2005) and in CSU research (Wassens 2006; Wassens and Amos 2011). Surveys were conducted in the periods June 2nd - 4th 2011 (pre-filling), July 5th - 7th 2011 (post-filling), and August 30th - September 2nd 2011 (post-flood 2), October 25th-30th 2011, December 14th - 18th 2011 and February 14th - 18th 2012 (Figure 4). Additional water quality samples were also collected on the peak flow (June 20-24th 2011).

### 2.1.2. Water quality changes in response to environmental watering

Water quality was monitored within each of the twelve wetlands and from three locations within the Murrumbidgee River: upstream of the wetlands (Murrumbidgee at Wagga Wagga), near to the wetlands (Murrumbidgee at Euroley) and downstream of most of the wetlands (Murrumbidgee at Darlington Point). During June 2011, water quality was also monitored at five in-stream sites upstream of Wagga Wagga. Field water quality variables (water temperature, conductivity, pH, turbidity, dissolved oxygen and water depth) were measured at each site using a handheld YSI meter.

Water samples were collected to assess for Total Organic Carbon (TOC), Dissolved Organic Carbon (DOC) and (by difference) Particulate Organic Carbon (POC), and nutrients on each of the field trips until the end of August 2011. Organic matter humic and fulvic character (by fluorescence and absorbance spectroscopy) was assessed on samples from all sites over this period (where access would allow) and continued at the wetland sites for the duration of the project.

Samples for dissolved organic carbon, soluble phosphate, ammonia, nitrate and fluorescence spectroscopy were filtered through a 0.45 $\mu$m membrane at the time of sampling. Unfiltered samples for total organic carbon and total nutrients and filtered samples for carbon and nutrients were frozen in the field and samples for fluorescence were stored on ice and analysed within one day of returning from the field. Fluorescence and absorbance scans were recorded on both acidified (HCl to give a final pH of approx 2) and non-acidifed samples from each site. Absorbance scans were recorded from 550 nm to 200 nm with a 1 nm step size. Fluorescence emission was recorded across this range using excitation wavelengths from 200 to 400 nm with a 10 nm step size. Fluorescence results were corrected for sample absorption and plotted as contour plots (Howitt, Baldwin *et al.* 2008). Nutrient analysis was undertaken by the Environmental Analytical Laboratories at Charles Sturt University and organic carbon analysis at the Murray Darling Freshwater Research Centre. Blanks were collected on the first sampling trip and took the form of ultrapure laboratory water, taken in the field and filtered in the same way as samples. Blanks were analysed for organic carbon and nutrients.

### 2.1.3. Vegetation

Biophysical variables (e.g. aquatic and fringing vegetation aquatic species and percent cover of each wetland plant species along with percent cover leaf litter and bare ground) were measured from within 1m$^2$ quadrats along three 30 m transects (90m$^2$ in total) starting at the higher waterline and running into the waterbody with measurement every 1 m.

### 2.1.4. Frogs

Adult frogs and metamorphs were surveyed at each wetland after dark using a 3 x 10 minute visual encounter and a 3 x 1 minute audio survey (Heyer, Donnelly *et al.* 1994). All individuals encountered were identified to species. Tadpoles and small-bodied fish were surveyed using timed 5 x 1 minute sweeps with soft mesh sweep-net in addition to 3 minute day time visual searches along the water's edge for egg masses and metamorphs.

### 2.1.5. Tadpoles and native fish

Intensive assessments of fish and tadpole communities were conducted at seven wetlands (Molleys, Dry Lake, Sunshower, Gooragool, Yarrada, Mckennas and Euroley (Control 1)). It was not possible to survey fish at either of the control 2 wetlands, Yanco Ag and Turkey flat because they were too shallow to set fyke nets (less than 50 cm). Intensive surveys of fish and tadpoles were conducted on four occasions August 30$^{th}$ to September 2$^{nd}$ , October 25$^{th}$-30$^{th}$, December 14$^{th}$-18$^{th}$ 2011 and February 14$^{th}$ to 18$^{th}$ 2012 using methodology previously described in (Spencer and Wassens, 2010). A combination of sampling methods targeting different habitats within each wetland were employed to survey for fish and tadpoles. Sweeping netting, seining, bait traps and large and small fyke nets were utilized to survey the seven intensive monitoring sites (Table 1). Five bait traps (dimensions 25 x 10 cm, 5 mm mesh), two large (2 x 10 m wings, 12 mm mesh) and two small (2 x 2 m wings, 2 mm mesh) fyke nets were left overnight at each site. Three replicate samples using a seine net and four small fykes were employed at Euroley wetland which was too shallow to sample using large fyke nets. Wing width and depth (m) were recorded at each site. Fish species were identified and measured to the nearest millimetre (standard length). Native fish were returned immediately to the water when possible, but alien fish were euthanased under NSW Fisheries ethics guidelines (NSW Fisheries approval ACEC 06/08). The percentage of recruits within the samples was estimated using size limits established by Gilligan (2005).

(a) Large Fyke Net

(b) Small Fyke Net

(c) Sweep Netting

(d) Bait Traps

Plate 3. Examples of fish and tadpole survey methods.

### 2.1.6. Waterbirds

Ground surveys for waterbirds were carried out in the mid-Murrumbidgee wetlands in June, July, August, October and December 2011 and February 2012. From August 2011 onwards, replicate ground counts were conducted over two consecutive days (one morning and one afternoon) to estimate maximum waterbird abundance and species diversity. Only single counts were completed during June and July 2011 surveys, and at Berry Jerry, Coonacoobil and Narrandera State Forest during each survey period. Birds were observed using binoculars (8 x 30 mm) and a telescope

(Swarovski 20 – 60X zoom). Total counts for each waterbird species and any evidence of breeding activity (nesting and/or the presence of young) were recorded during each survey.

### 2.1.7. Wetlands data analysis

Waterbird species richness and abundance in the wetlands were compared with Analysis of Variance (ANOVA) (Systat 2007). Data were log transformed to improve normality and stabilise variance. Multivariate analyses (PRIMER 2002) were used to investigate differences in species assemblages among the wetlands. Data were fourth root transformed to control for multiple zeros and large values present in the data sets (Quinn and Keough 2002). The transformed abundance data were examined using the Bray-Curtis measure of similarity (Bray and Curtis 1957) and one-way Analysis of Similarity (ANOSIM) was used to detect significant differences in species assemblages among the wetlands. Analysis of Similarity Percentages (SIMPER) was used to determine the contribution made by each species (Clarke and Warwick 2001).

## 2.2. Monitoring of in-stream parameters

This project monitored biofilms and macroinvertebrates to assess the instream ecological response to the environmental flow event in the Murrumbidgee River in June 2011. The following hypotheses were examined in this study:

- Algal and total biomass on cobbles will decrease following the environmental flow pulse due to scouring of biofilms from increased flow velocity.
- Following the environmental flow pulse, there will be a change in the community composition of algal biofilms, with an increase in early successional algal taxa due to scouring from increased velocity.
- There will be an increase in the diversity of aquatic macroinvertebrates following the environmental flow pulse.

- Following the flow pulse there will be a shift in community composition of aquatic macroinvertebrates towards the composition observed in the unregulated tributary.

### 2.2.1. Discharge data

Daily discharge data were obtained from NSW Government water information website (NSW Office of Water, 2012) for gauging stations in the Murrumbidgee River downstream of Burrinjuck Dam at Glendale (410068), Tumut River downstream of Blowering Dam at Oddys Bridge (410073), and the Goobarragandra River at Lacmalac (410057), which is an unregulated tributary of the Tumut River.

Severe drought conditions occurred throughout eastern Australia from 2000-2010. The period from February 2006 to September 2010 was extremely dry and releases from Burrinjuck Dam to the Murrumbidgee River during this period were less than 4,000 ML/day. During the same period releases from Blowering Dam to the Tumut River were regulated for consumptive use, with the discharge often being between 6000 ML/d and 8000 ML/day from late spring to early autumn.

A number of large natural flow events occurred between September 2010 and January 2011 coinciding with heavy rainfall in the region. In December 2010 Burrinjuck Dam spilled resulting in a discharge of 142,030 ML/day at the Glendale gauge downstream of the dam (Figure 5).

The water for the June 2011 environmental watering event was released by State Water Corporation from both Burrinjuck and Blowering Dams. The release commenced on June 14$^{th}$ and reached maximum daily discharge of 24,908 ML/day in the Murrumbidgee River downstream of Burrinjuck Dam on 17$^{th}$ June 2011 and 9,492 ML/day in the Tumut River downstream of Blowering Dam on 16$^{th}$ June 2011 (Figure 6). Following the environmental watering event there were two smaller flow pulses of different magnitudes from Burrinjuck Dam in mid August and early September. During the study period there were several regulated releases from Blowering Dam reaching a maximum discharge of approximately 8000 ML/day and one event in July continued at this discharge for approximately three weeks.

In general, the discharge pattern from Burrinjuck Dam during the six years from January 2005 to December 2011 was more similar to the flow regime in the unregulated Goobarragandra River than to the regulated regime in the Tumut River downstream of Blowering Dam (Figure 5).

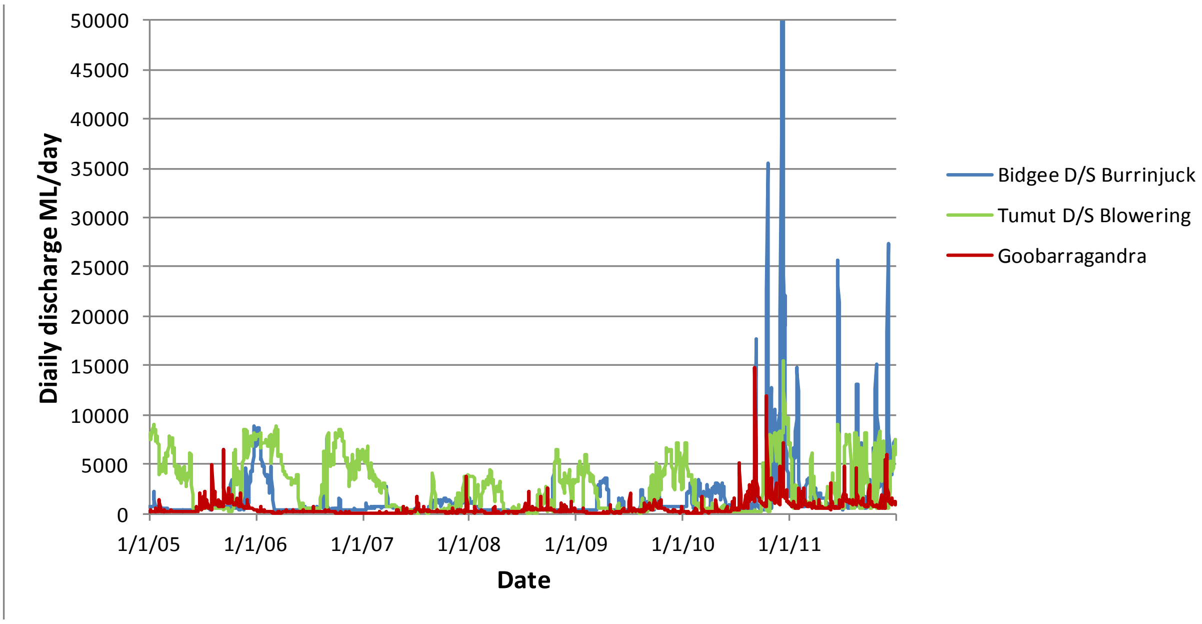



Figure 5. Hydrograph showing daily discharge in the Murrumbidgee River downstream of Burrinjuck Dam, Tumut River downstream of Blowering Dam and in the Goobarragandra River (unregulated river) for the period from Jan 2005 to December 2011. Burrinjuck Dam spilled in December 2010 resulting in a discharge of 142,030ML/day at the Glendale gauge downstream of the dam (values truncated in this figure).

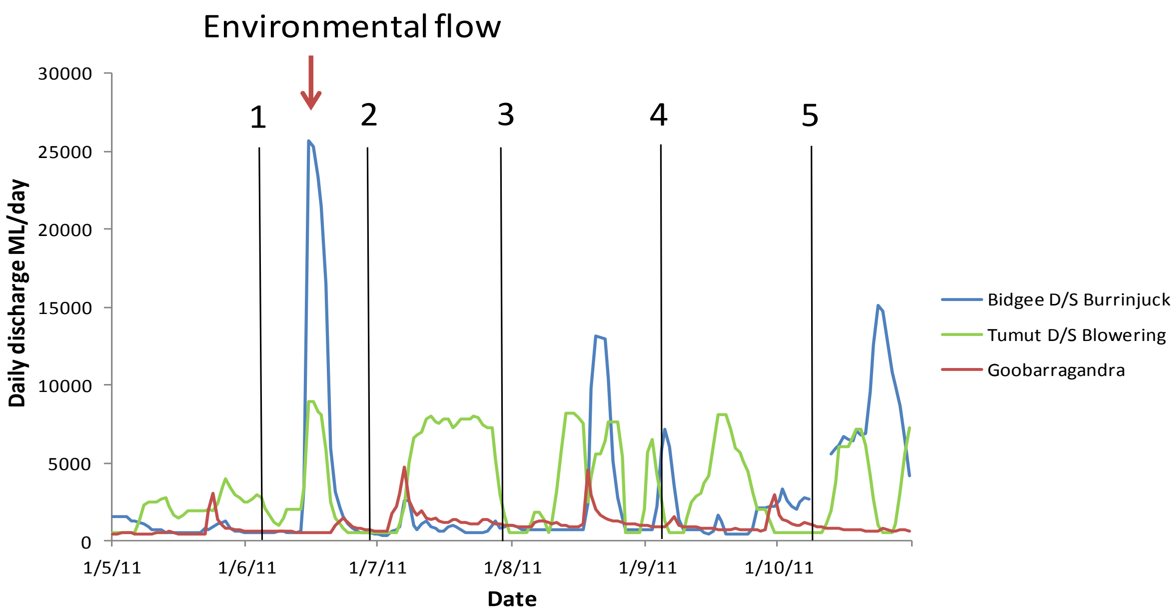



Figure 6. Timing of survey events in relation to discharge in the Murrumbidgee River downstream of Burrinjuck Dam, Tumut River downstream of Blowering Dam and in the Goobarragandra River (reference stream). Lines indicate timing of (1) survey in June 2011 prior to the environmental flow, and (2-5) for surveys conducted after the environmental flow.

### 2.2.2. Field sampling

Monitoring of in-stream parameters was undertaken in three reaches of the Murrumbidgee River downstream of Burrinjuck Dam and four reaches of the Tumut River downstream of Blowering Dam (Figure 7). One reach in the Goobarragandra River, an unregulated tributary of the Tumut River, was sampled as a reference. Each of the river reaches selected for the study included a cobble bench (Plate 4), a slow flowing pool habitat and sections of vegetated edge habitat.

Plate 4. Murrumbidgee River at Gundagai (left) and Jugiong (right) showing presence of cobble benches at each site.

Sampling was undertaken on five survey dates; once prior to the environmental flow pulse in late June 2011, and four times after the environmental flow (Figure 6). The fourth sample from Murrumbidgee site 1 was collected on the 9$^{th}$ September (five days later than all other sites were sampled) because a flow release from Burrinjuck Dam on the morning of 4th September prevented this site from being sampled safely for several days until the flow had receded.

A Horiba water quality monitor was placed just below the water surface at each site on each sampling date to obtain spot measures of the temperature (ºC), specific conductivity (mS/cm), dissolved oxygen (%), pH, and turbidity (NTU) of the water.

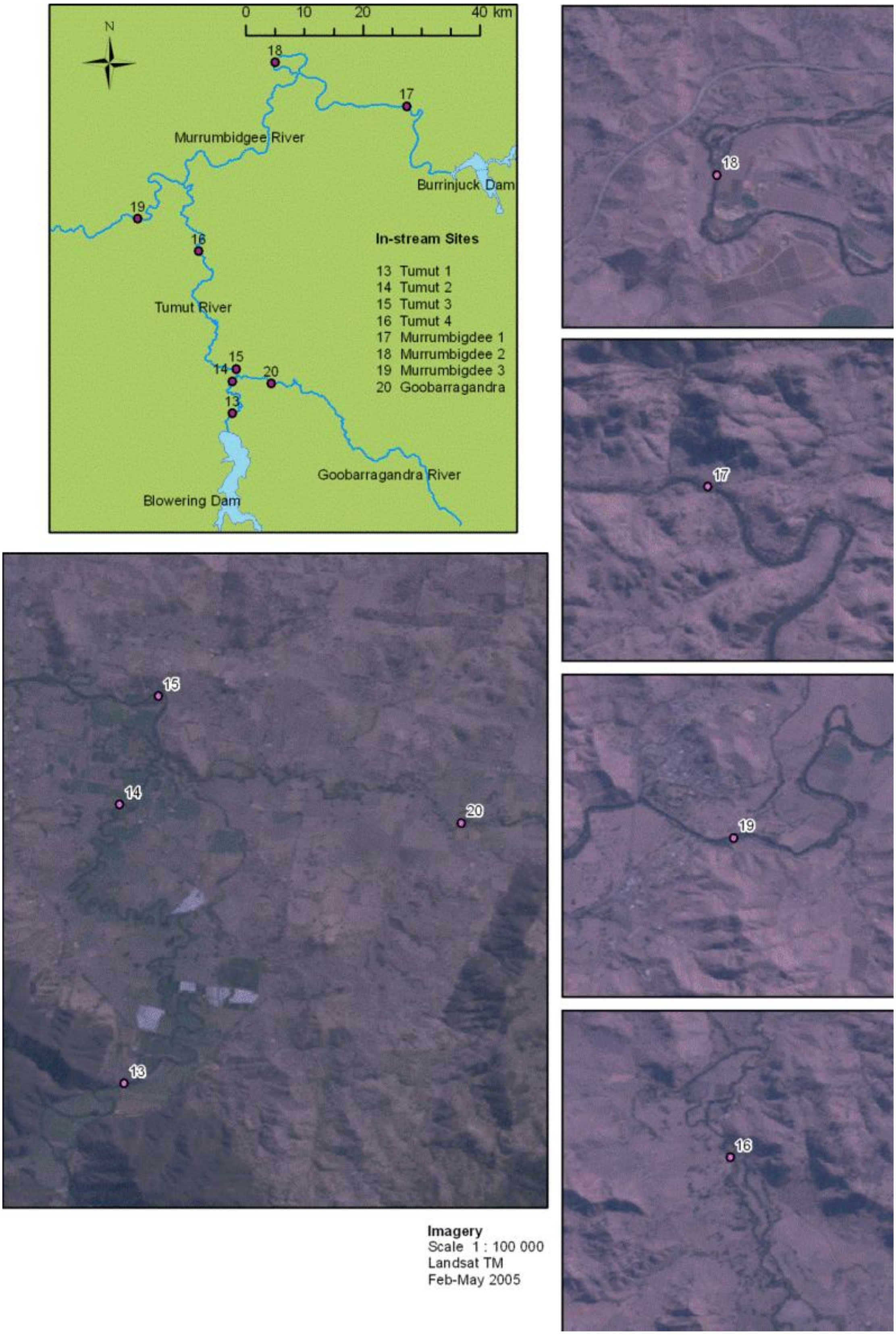



Figure 7. Distribution of in-stream monitoring sites. LandSat derived imagery provides individual site context.

An area in the channel at each river reach that would remain permanently inundated throughout the study period was selected for sampling biofilms. On each sampling occasion five cobbles (ranging between 10 and 20cm diameter) were collected from this area within each reach, placed in labelled sealed plastic bags and stored in the dark in an esky for transport back to the laboratory.

Aquatic macroinvertebrates were sampled from riffle, pool and edge habitats at each river reach using a sweep net with 250 micrometre mesh. Within each habitat a total of five metres was sampled (sometimes made up of a composite of smaller length to a total of five metres); the riffle and pool habitats were sampled using a kick sample method, and the pool sampled using a sweep sampling method. Following sampling, net contents were emptied into a labelled sample jar and were preserved in 70% alcohol.

### 2.2.3. Laboratory processing of samples

The biofilm was scrubbed from each cobble into 100 millilitres of distilled water using a soft nailbrush within 12 hours of field collection. Sub-samples were removed from the 100 millilitre residue for determination of chlorophyll-a and were filtered through a GC-50 0.5 micrometres filter. The amount filtered was recorded and a 10 millilitres sample for the assessment of taxonomic composition was stored in Lugols solution. Using GC-50 0.5 micrometres filter papers (for which the loss on ashing had been predetermined) a recorded amount of the solution was filtered, the 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). Chlorophyll-*a* was determined following (Tett, Kelly *et al.* 1975). Samples were placed in 8 millilitres of 90% methanol containing 150 milligrams magnesium hydroxide carbonate, extracted for 18 hours at 4°C, transferred to a 70°C water bath and boiled for two minutes. Samples were centrifuged at 4500 rpm for three minutes and optical densities at 750 and 666 nanometres were measured pre- and post-acidification (1M HCl) using a UV/Visible Spectrophotometer.

Each cobble was measured for colonisable rock surface area (CRSA) by covering the exposed surface area of the rock (excluding the buried surface) with aluminium foil after (Doeg and Lake 1981). CRSA measurements were used to standardise biofilm dry weight (DW) and ash free dry weight (AFDW) to $g/m^2$ and chlorophyll-*a* to $mg/m^2$. Percent organic matter was calculated as the proportion of AFDW to DW and converted to a percentage to standardise across sites and dates.

Identification of the macroinvertebrate samples was undertaken using a dissecting microscope. Fauna were identified to family level with the exception of mites, flatworms, nematodes and oligochaetes. The number of taxa and number of individuals were calculated for each habitat and for each site.

# 3. RESULTS

## 3.1. Mid-Murrumbidgee wetlands

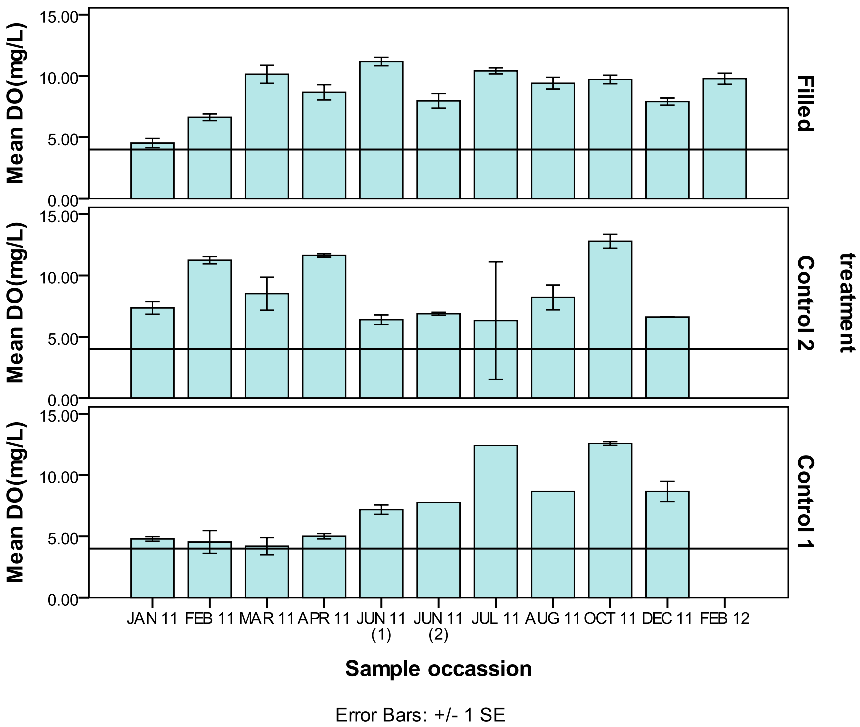
Plate 5. Gooragool Lagoon February 2012.

### 3.1.1. Water quality

***Dissolved Oxygen***

Dissolved Oxygen concentrations were usually well above 4mg/L which is the threshold of concern for aquatic health (ANZECC water quality guidelines 2000) throughout the study within all treatment types (Figure 8). The exception to this was a single very low measurement at Yanco Ag in July, where very shallow water increased the risk that any disturbance of anaerobic sediment during sampling may have decreased the measured dissolved oxygen concentration. Overall DO concentrations were similar between the three treatments ($F=2.148$, $p = 0.119$). As expected DO levels changed significantly over time but the pattern of change varied between treatments ($F = 6.351$, $p <0.001$). DO concentrations within filled wetlands exceeded 8mg/L by March 2011 and remained relatively stable from that point onwards which indicated that the environmental releases in June and December did not have an overall significant impact on DO levels. In contrast DO levels within control 2 wetlands were variable over time, probably because they received multiple small inflows during the study period. The control 1 wetlands had low DO during the first survey year, but increased by June 2011 and remained at levels similar to the filled wetlands until drying out in January 2012.

Water temperature influences the solubility of oxygen in water as less oxygen can dissolve in warm water than in cold water. It is also useful to consider the dissolved oxygen concentrations expressed as percentage saturation (DO%). Less than 100% DO indicates that biological or chemical processes in the water are consuming oxygen faster than it can be replenished from the air. Values greater than 100% indicate that photosynthesis by algae or plants is producing oxygen in the water column faster than it can escape to the atmosphere or be consumed by other organisms. Guidelines for dissolved oxygen in lowland rivers of south-east Australia suggest that the DO% saturation should lie between 85-110% (ANZECC 2000). Separate guidelines are not available for wetland ecosystems. The oxygen concentrations were below the 85% level in control 1 wetlands from Jan 2011 until August 2011 and within Control 2 from June 2011 until October 2011. DO % levels briefly dropped below the 85% level within filled wetlands (and some river sites) immediately during filling but had recovered by August, however based on the DO mg/L levels this drop is not considered sufficient to have contributed to declines in fish health within wetlands. It is noted that Coonacoocabil wetland exhibited a further decrease in oxygen saturation in August during the natural flow pulse and then rebounded back to super-saturated conditions by October (Appendix 1).



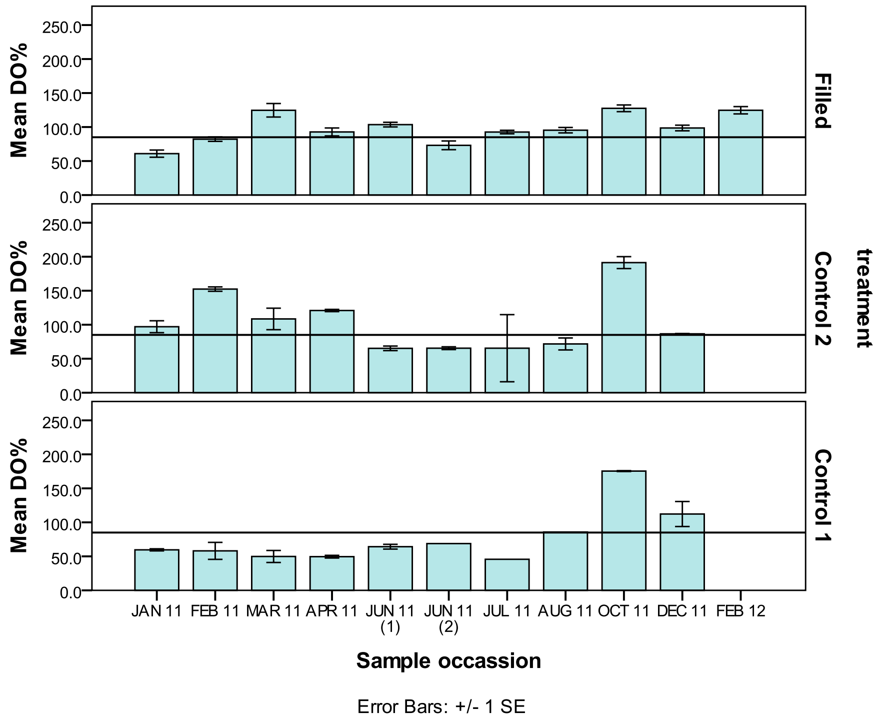




Figure 8. Mean (± SE) dissolved oxygen levels (DO) mg/L (top) and dissolved oxygen % (DO%) (bottom) within wetlands in each treatment type over time (January 2011-April 2011 from Wassens and Amos, 2011, and June 2011-Febuary 2012 from current monitoring program). The first environmental release occurred after the June 2011 (1) survey. Note that control wetlands were dry during surveys in February 2012.

### *Conductivity*

Conductivity levels were within the normal acceptable range for lowland rivers (0.015-0.200 mS/cm) (ANZECC water quality guidelines 2000) within filled and Control 1 wetlands (Figure 9). Conductivity levels were significantly higher within Control 2 wetlands (F = 105.34, p <0.001), which were above the normal acceptable range. Conductivity levels changed over time, the environmental release in June 2011 lowered conductivity within the filled wetlands which then slowly increased towards the pre filling levels in December 2011 and February 2012 as wetlands started to dry. Conductivity within Control 2 wetlands were extremely variable over time which reflects the pattern of frequent low level run off entering these wetlands from the surrounding area.

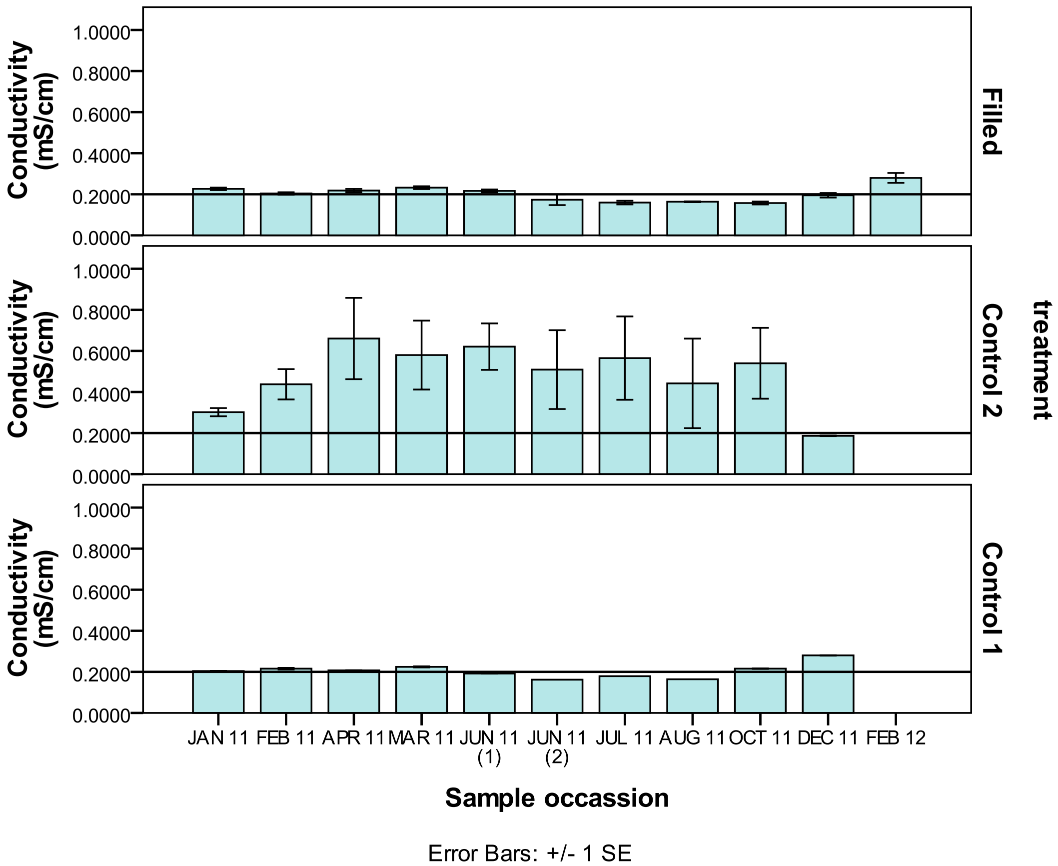



Figure 9. Mean (± SE) Conductivity $mScm^{-1}$ within wetlands in each treatment type over time (January 2011-April 2011 from Wassens and Amos, 2011, and June 2011-Febuary 2012 from current monitoring program). The first environmental release occurred after the June 2011 (1) survey. Note that control wetlands were dry during surveys in February 2012.

### *pH*

pH levels were similar between the three treatments (F = 2.12, p =0.122) and were within the normal acceptable levels for lowland rivers (no trigger levels given for wetlands) (ANZECC water quality guidelines 2000) (Figure 10). There were minor changes over time within the filled and control 1 wetlands but no clear trends were evident. pH levels within control 2 wetlands were more variable over time than in either filled and control 1 wetlands (F= 4.16, p <0.001).

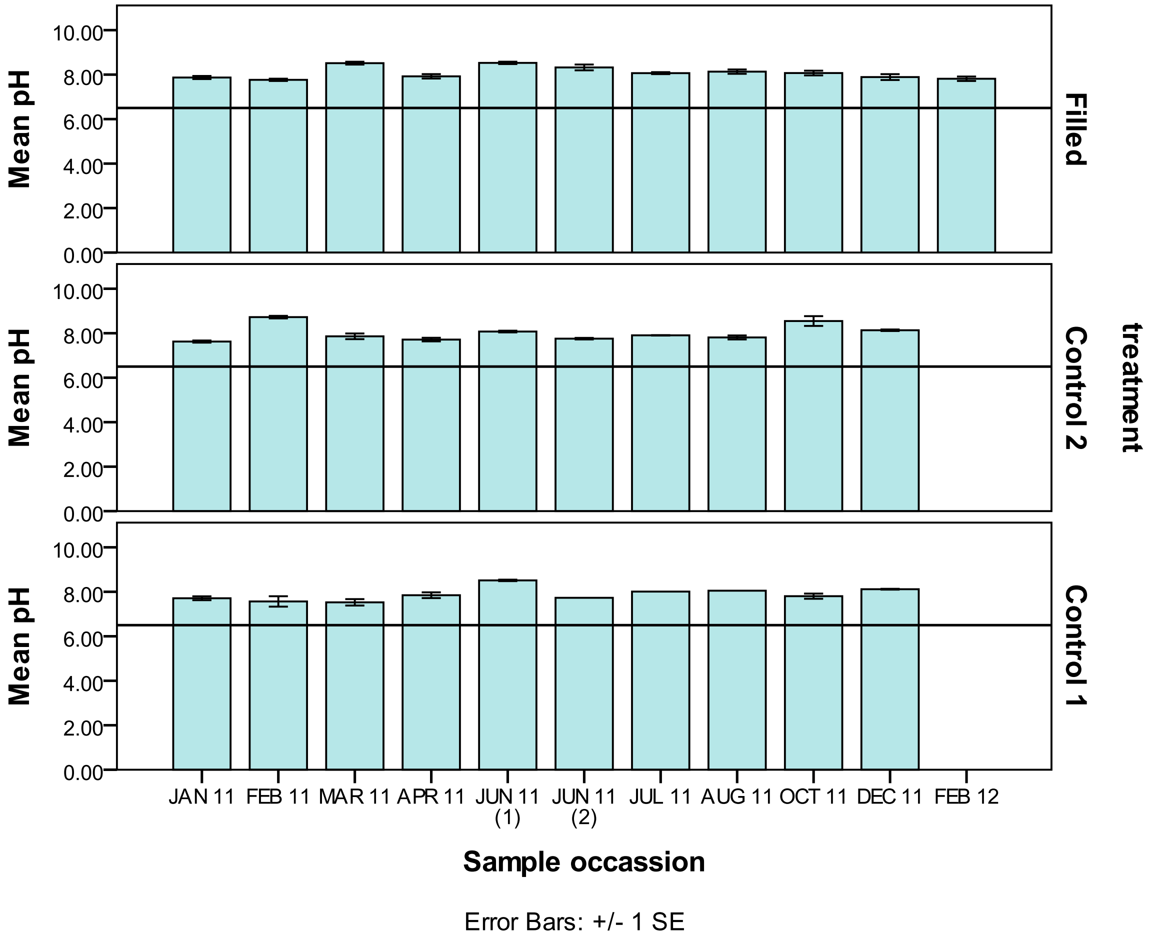



Figure 10. Mean (± SE) pH within wetlands in each treatment type over time (January 2011-April 2011 from Wassens and Amos, 2011 and June 2011-Febuary 2012 from current monitoring program). The first environmental release occurred after the June 2011 (1) survey. Note that control wetlands were dry during surveys in February 2012.

***Water temperature***

As expected water temperature was similar between the controls and the filled wetlands and varied over time reflecting seasonal temperature changes (Figure 11). Water temperatures were very low at the time of the environmental release, June 2011(2), and did not increase until August 2011.

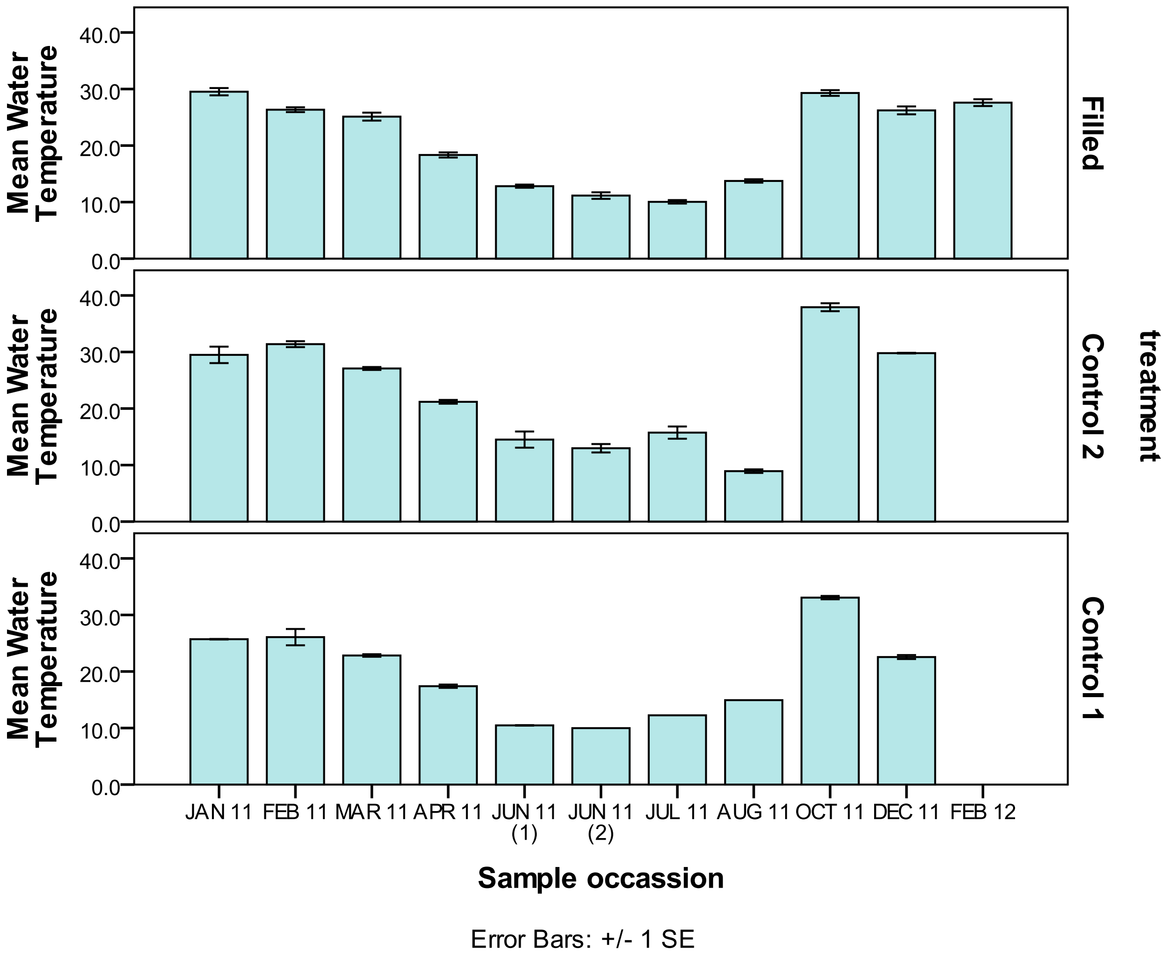



Figure 11**.** Mean (± SE) water temperature (°C) within wetlands in each treatment type over time (January 2011-April 2011 from Wassens and Amos, 2011 and June 2011-Febuary 2012 from current monitoring program). The first environmental release occurred after the June 2011 (1) survey. Note that control wetlands were dry during surveys in February 2012.

### *Turbidity*

Turbidity levels were similar between the three treatments (F = 1.62, p = 0.119) but changed over time (F = 7.30, p <0.001), increasing towards the end of the monitoring period as wetlands began to dry out and sediments became resuspended within the wetlands due to wind action (Figure 12).

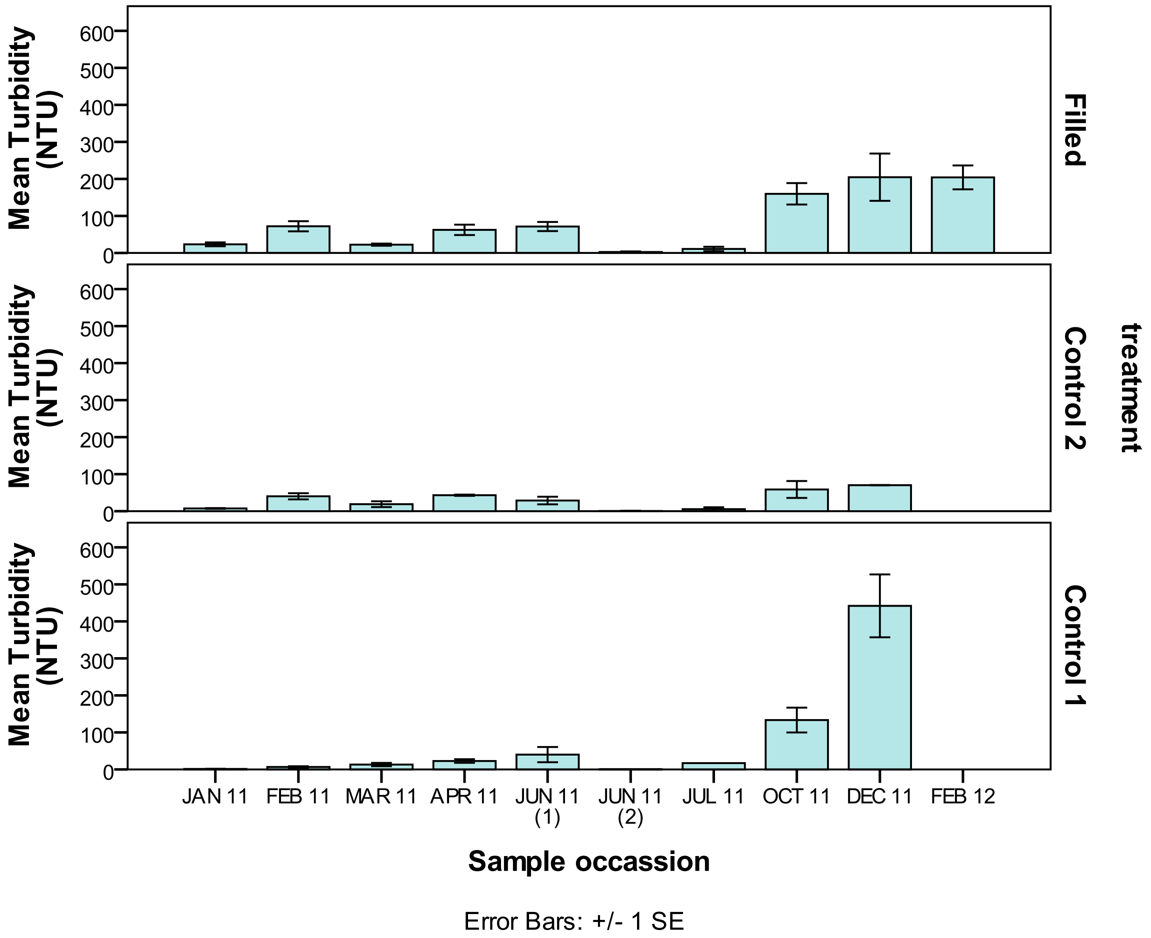



Figure 12**.** Mean (± SE) turbidity (NTU) within wetlands in each treatment type over time (January 2011-April 2011 from Wassens and Amos, 2011, and June 2011-Febuary 2012 from current monitoring program). The first environmental release occurred after the June 2011 (1) survey. Note that the control wetlands were dry during surveys in February 2012.

### *Carbon*

Dissolved organic carbon (DOC), total organic carbon (TOC) and particulate organic carbon (POC) were measured within the filled and control 1 and 2 wetlands, the Murrumbidgee River (downstream of Burrinjuck Dam) to Darlington Point, in the Tumut River downstream of Blowering Dam and the Goobarragandra River (see maps in Figure 3 and Figure 7), before the environmental release in early June, during the peak of the release in late June, in early July after the peak and again in late August.

Prior to the environmental release the total organic carbon (TOC) and dissolved organic carbon (DOC) concentrations in both filled and control wetlands were relatively high (10 to 40 mg/L) compared to the river sites which were always less than 10 mg/L. DOC, TOC and POC concentrations decreased following the environmental release at all wetlands except for McKennas and Dry Lake, where DOC and TOC levels remained high following the initial filling event and declined more slowly over time (Figure 13). By August however the DOC, TOC and POC concentrations had declined at all of the filled wetlands to levels only slightly higher than the Murrumbidgee between Wagga Wagga and Darlington Point. In contrast TOC, DOC and POC concentrations at Euroley (C1), which did not receive any inflows during the study period, increased over time with POC concentrations almost doubling between June and August.

Further analysis of the organic carbon composition in the wetlands was undertaken using absorbance and fluorescence spectroscopy (see Appendix 2) and indicates that not only the amount of dissolved organic carbon changed over time, but also the composition and likely bioavailability were influenced by reconnection between the wetlands and the river.

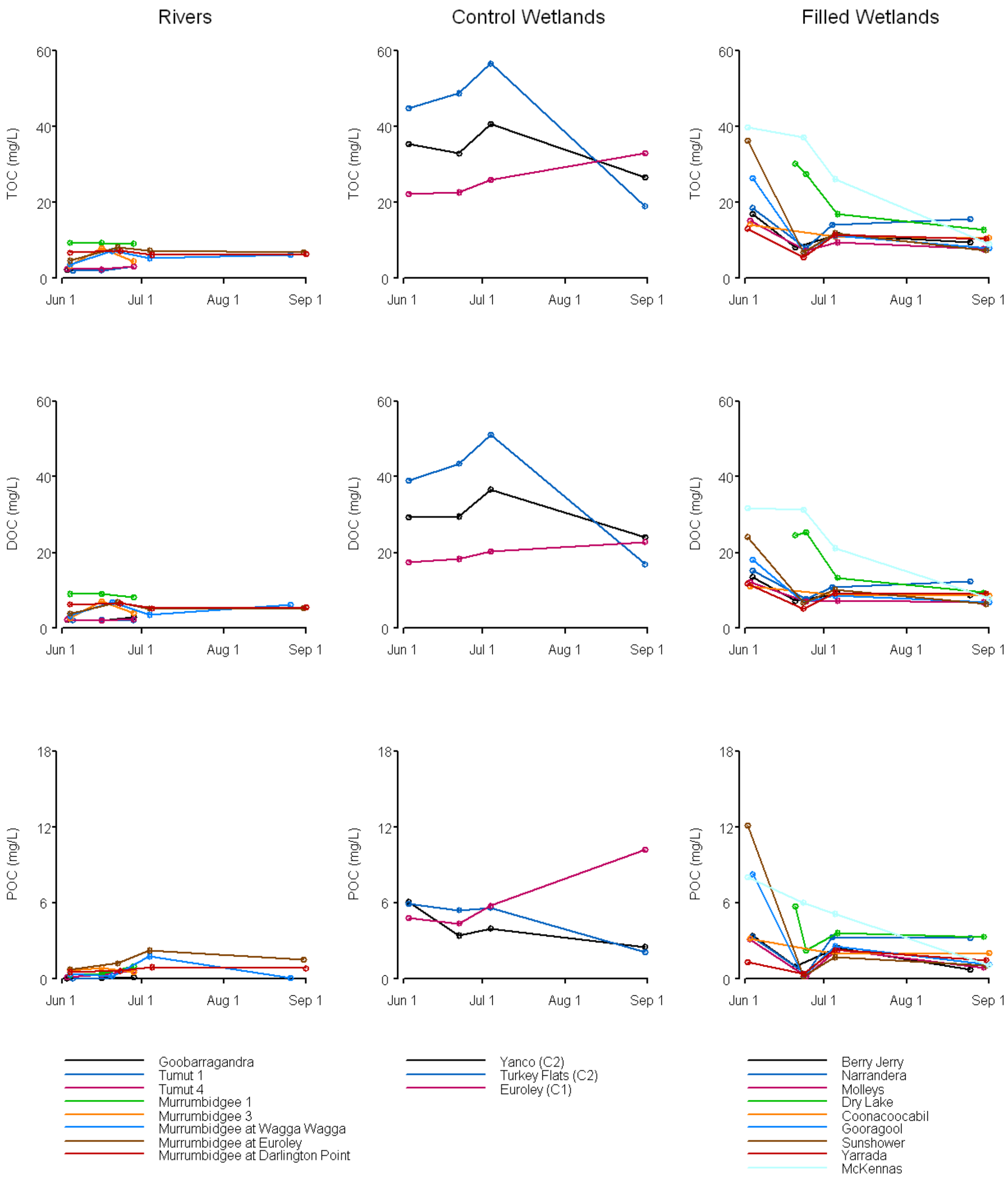



Figure 13. Total organic carbon (TOC), dissolved organic carbon (DOC) and particulate organic carbon (POC) at in-stream, control wetlands treatment (filled wetlands) before, during and after the June 2011 environmental water release. Note that the y axis of the POC graphs is presented on a smaller scale for clarity.

***Nutrients***

Nutrient analyses were undertaken between June and August 2011. Four parameters were measured: Ammonia, Nitrate and Nitrite ($NO_x$), Total Kjeldahl Nitrogen and Phosphate. When wetlands are dry they can store large amounts of nutrients and carbon. These are released following inundation which means that nutrient levels in wetlands are naturally higher than in the river. At present the ANZECC guidelines for water quality do not set trigger levels for nutrients within floodplain wetlands so it is not possible to assess nutrient levels against these criteria. Instead the data presented here are discussed in the context of levels recorded post-inundation within river red gum wetlands in the Macquarie Marshes (Kobayashi, Ryder *et al.* 2009).

Overall the levels of ammonia were below the limit of detection within filled wetlands throughout the study. Similar results have been recorded for river red gum wetlands in the Macquarie Marshes (Kobayashi, Ryder *et al.* 2009) (see appendix 3 for full nutrient data). The Control 2 wetlands had comparatively higher levels of Ammonia but these were within the same range as recorded for canopy sites within the Macquarie Marshes. Concentrations of $NO_x$ (combined nitrate and nitrite) were typically below the limit of detection in all wetlands, the one exception being the flood peak period when very low levels were recorded within the river and the filled wetlands, however these levels were well below those recorded in similar wetlands within the Macquarie Marshes (Kobayashi, Ryder *et al.* 2009). Total Kjeldahl Nitrogen levels were typically below the limit of detection within filled wetlands but were higher within the control 2 wetlands which receive run-off from the surrounding agricultural areas. Despite being higher within controls Total Kjeldahl Nitrogen were still within a similar range as canopy wetlands within the Macquarie Marshes.

Total dissolved phosphorus levels were typically higher within the control 2 wetlands than either the river or the filled wetlands. In the period after the flow peak the dissolved phosphorus levels increased within the river and filled wetlands ranging between 0.04 and 0.07 mg/L. These were still considerably lower than levels recorded within the Macquarie Marshes after six days of inundation which ranged from 0.5 mg/L to 1 mg/L (Kobayashi, Ryder *et al.* 2009).

## Vegetation communities

Vegetation communities were surveyed once each survey occasion between June 2011 (prior to release) and February 2012 as part of this study (2011-12). These data have been added to an existing dataset for the previous year (November 2010 to April 2011) (2010-11) (Wassens and Amos, 2011) in order to describe longer term trends in vegetation recovery. A non-parametric ANOVA equivalent (ANOSIM) (PRIMER version 5) is used to test for statistically significant differences in the community composition (Percent cover and composition of species) between the three treatments (Filled (received environmental water), control 1 (not filled during environmental release), control 2 (not filled during environmental release, received water from other sources) and changes over time.

### Key trends

- The diversity and percent cover of aquatic vegetation species increased significantly over time within the filled wetlands but not within the control wetlands (they supported similar aquatic species throughout the study).

- Control 2 wetlands had similar percent cover of aquatic vegetation between the 2010-11 and 2011-12 flood years, while filled wetlands showed a significant increase in percent cover of aquatic vegetation in year two. Control 1 wetlands had low aquatic cover in both years.

- Control 1 wetlands had a higher proportion of terrestrial vegetation species compared with filled and control 2 wetlands

- The recovery of aquatic vegetation was assessed in terms of flooding history within filled wetlands, overall wetlands that had been dry for between three and five years had higher rates of recovery of aquatic vegetation than those which had been dry for more than five years.

### 3.1.2. Vegetation communities

Aquatic vegetation communities differed significantly among wetlands (ANOSIM Global R 0.41, p <0.001), reflecting the wide diversity of wetland types and flooding histories. The aquatic vegetation communities differed significantly between control 1 and filled wetlands (R = 0.38, p =0.047), and between the control 1 and control 2 wetlands (R = 0.69, p = 0.001). Overall there were no significant differences in aquatic vegetation community compositions between the filled wetlands and Control 2 (wetlands flooded frequently via rainfall/irrigation diversion) and they both supported similar aquatic vegetation communities by the end of the study.

The composition of vegetation communities (type and percent cover of species) changed significantly over time within the filled wetlands (ANOSIM Global R 0.13, p= 0.04) but not within the control 1 or 2 wetlands (ANOSIM Global R -0.02, p= 0.537). Control 2 wetlands which had regular low level filling through rainfall and run-off had a similar composition and percent cover of aquatic vegetation between the 2010-2011 and 2011-2012 survey years. In order to present a simplified representation of the change to vegetation covers in control 1, 2 and filled wetlands vegetation species have been classified as being either terrestrial, semi-aquatic or aquatic and the mean percent cover within each treatment type is presented (Figure 14). Control 1 wetlands had limited aquatic vegetation in both years and were dominated by terrestrial species in 2011-12. In contrast the filled wetlands had initially very low percent cover of aquatic and semi-aquatic vegetation in 2010-11 and cover increased in these wetlands following the environmental water releases in 2011-12.

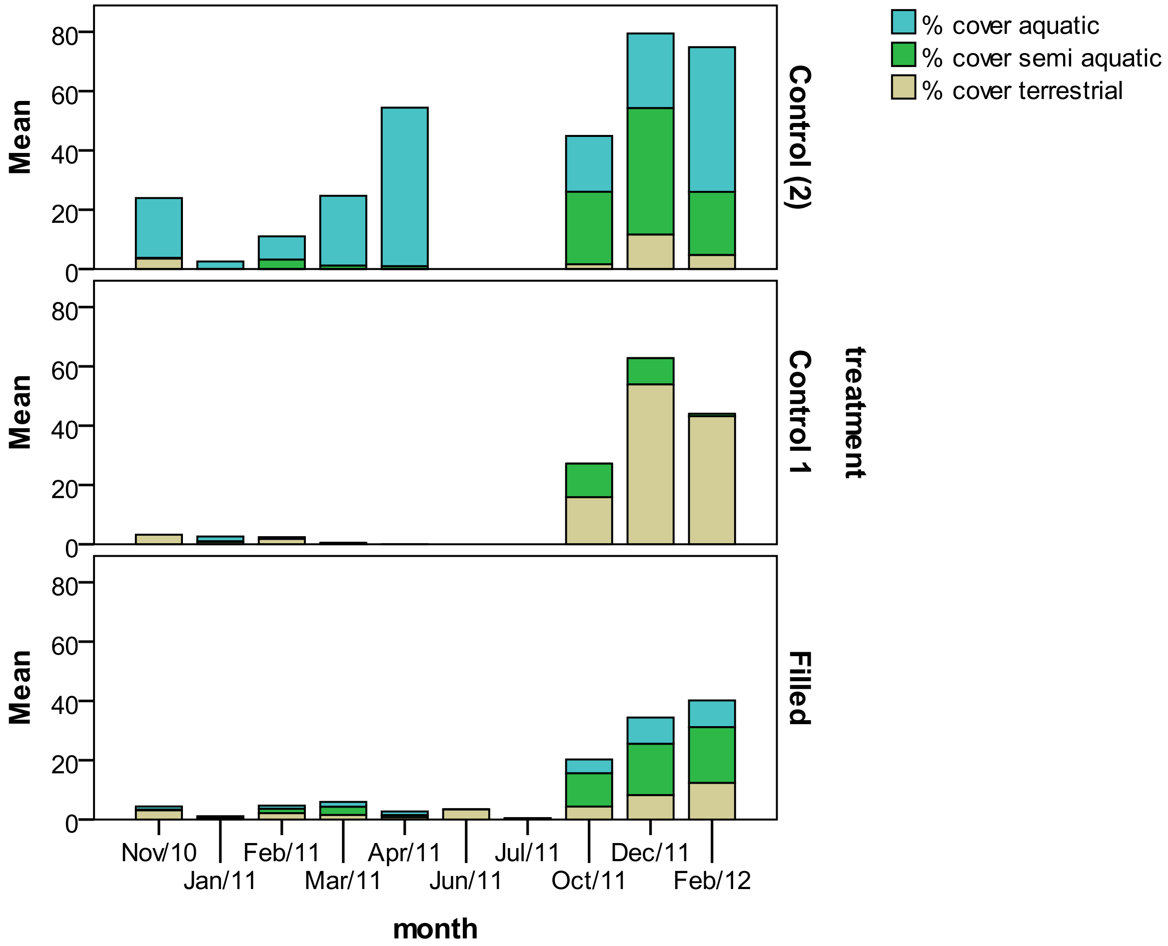



Figure 14. Aquatic, semi-aquatic and terrestrial vegetation cover at the control and filled wetlands across the period of study.

### *Factors influencing vegetation responses within filled wetlands*

In this section we are only considering wetlands that received water as part of the June 2011 environmental releases with a focus on describing the relationships between past flooding history (number of years dry) and vegetation response. The change in vegetation cover is considered with respect to history of inundation of the filled wetlands (bearing in mind that the majority of wetlands in this study had been dry for periods that exceeded their normal long-term filling regimes). The change in vegetation cover is a measure of the extent of vegetation recovery following wetting.

The recovery of aquatic vegetation was strongly linked to the length of time that the wetlands had remained dry (Jonckheere-Terpstra test for ordered alternatives Z -2.33, p = 0.02). For example, aquatic vegetation cover increased by an average of 25% from November 2010-February 2012 within wetlands that had been dry for between three and five years but by less than 7% within wetlands that were dry from between six and eight years (Figure 15). However, semi-aquatic vegetation species had a similar response following flooding regardless of flooding history (Z = -1.58, p = 0.114).

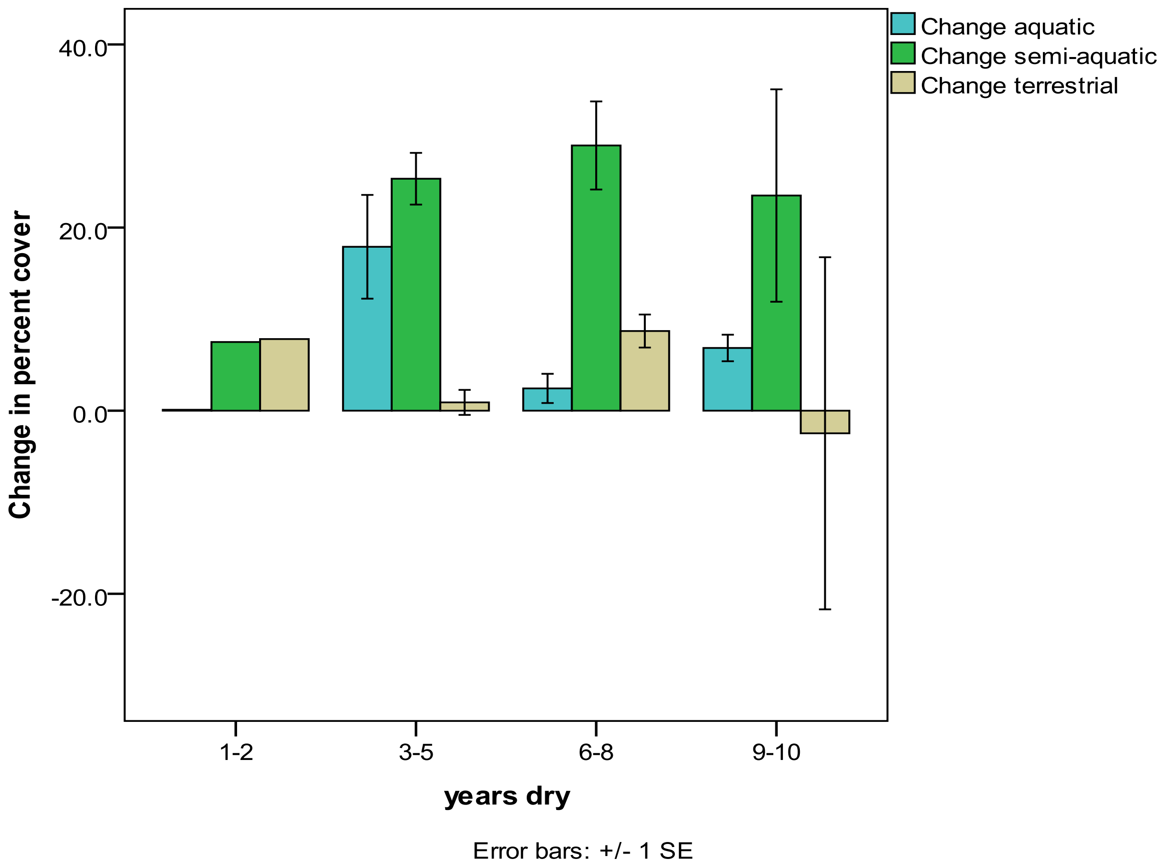



Figure 15. Change in aquatic, semi-aquatic and terrestrial vegetation cover organised by preceding number of years wetland was dry.

The clearest example of slow recovery of aquatic vegetation following prolonged drying was at McKennas lagoon (Plate 6) (dry eight years) which historically supported dense stands of tall spike rush *Eleocharis sphacelata* and had a very high percent cover of aquatic vegetation. After eight years without water the aquatic vegetation communities did not recover following the environmental releases to the same extent as those wetlands which had been dry for between three and five years such as Gooragool Lagoon (Plate 7). Tall spike rush which was abundant at McKennas Lagoon in 2001 was not recorded during our surveys in 2010-12.

Plate 6. McKennas Lagoon in 2001 (James Maguire NSW OEH) (left) and in February 2012 (right).

Plate 7. Gooragool Lagoon in June 2011 (pre-filling) (left) and February 2012 (right). Arrows show common points on the image.

# Frogs

Frog communities were surveyed once each survey occasion between June 2011 (prior to release) and February 2012 as part of this study (2011-12). These data have been added to an existing dataset for the previous year (November 2010 to April 2011) (2010-11) (Wassens and Amos, 2011) in order to describe longer term trends in the recovery of frog communities. Mann-Whitney U test was used to test for significant differences in the abundance of each frog species over time within each of the three treatments (**filled** (received environmental water), **control 1** (not filled during environmental release), **control 2** (not filled during environmental release, received water from other sources) and changes over time.

## Key trends

- Five species of frog were recorded between June 2011 and February 2012.

- Tadpoles of barking marsh frog, spotted marsh frog and Perrons tree frog were recorded in both filled and control 2 wetlands; no tadpoles were recorded in the control 1 wetland.

- The abundance of barking marsh frogs and spotted marsh frogs increased significantly between the 2010-11 and 2011-12 water years within filled wetlands, but not within any of the control 1 or 2 wetlands.

- Tadpoles of barking marsh frogs and spotted marsh frogs were most abundant in October 2011, fewer tadpoles were recorded in December 2011 and February 2012 and these were typically of a later development stage (closer to undergoing metamorphosis).

### 3.1.3 Frogs

Five species of frogs were recorded between June 2011 and February 2012 (Table 2). Four species, the plains froglet *Crinia parinsignifera*, Perons tree frog *Litoria peronii*, barking marsh frog *Limnodynastes fletcheri* and spotted marsh frog *Limnodynastes tasmaniensis*, were widespread occurring at all of the filled and control sites. The inland banjo frog *Limnodynastes interioris* was restricted to a single site, with one juvenile frog recorded at Molleys lagoon in August 2011 (Plate 1). We did not record Southern bell frogs *Litoria raniformis* at Sunshower or Gooragool lagoons where they had previously been recorded in November 2010.

Tadpoles were recorded at five of the filled wetland sites and in both of the control 2 (Yanco Ag and Turkey Flat) sites with the majority of tadpoles recorded within the control 2 wetlands in February 2012. No tadpoles were recorded in Euroley (control 1).

Table 2. Frog species recorded at mid-Murrumbidgee wetland sites from June 2011 to February 2012

| | Plains froglet | Barking marsh frog | Inland banjo frog | Peron's tree frog | Spotted marsh frog | Perons tree frog | Spotted Marsh frog/barking marsh frog |
|---|---|---|---|---|---|---|---|
| | Frogs | | | | | Tadpoles | |
| Berry Jerry | * | * | | * | * | | |
| Coonancoocabil | * | * | | * | * | | |
| Dry Lake | * | * | | * | * | | * |
| Gooragool | * | * | | * | * | | * |
| McKennas | * | * | | * | * | * | * |
| Molleys | * | * | * | * | * | | |
| Narrandera SF | * | * | | * | * | * | |
| Sunshower | * | * | | * | * | | * |
| Yarrada | * | * | | * | * | | * |
| Euroley (C1) | * | * | | * | * | | |
| Turkey Flats (C2) | * | * | | * | * | | * |
| Yanco Ag (C2) | * | * | | * | * | * | * |

### *Change in frog populations*

The abundance of each frog species across the 12 survey wetlands was compared between the 2010-11 (November 2010 – April 2011) survey years (Wassens and Amos, 2011) and June 2011- February 2012 (current surveys) using a non-parametric equivalent to ANOVA (Mann-Whitney U test) in order to describe trends in frog populations over time. The abundance of each species of the five most commonly recorded species was compared for wetlands in their first year of flooding (when they filled for the first time after an extended dry period following a natural flood event see (Wassens and Amos, 2011) and during the 2011-12 environmental release using the Wilcoxon–Mann–Whitney two-sample rank-sum test (Table 3). The abundance of frogs in wetlands that received water during the 2011-12 environmental releases was generally higher than in the two controls (which did not receive water as part of the environmental release), but outcomes differed among species (Figure 16). Abundance of barking and spotted marsh frogs increased significantly within wetlands that received environmental water in 2011-12, but not in either of the control wetlands. In contrast, the abundance of Peron's tree frogs and plains froglet did not differ significantly between the two years in wetlands that received environmental water or the controls.

Table 3. Mann-Whitney U comparison in the abundance of each frog species within the three treatments (filled, control 1 and control 2 wetlands) across the two survey years. The inland banjo frog has been excluded due to small sample sizes (n = 3).

| Common name | | Filled | Control 1 | Control 2 |
|---|---|---|---|---|
| **Barking marsh frog** | Mann-Whitney U | 574.50 | 0.00 | 87.50 |
| | Z | -2.18 | -1.00 | -0.12 |
| | Asymp. Sig. (2-tailed) | **0.03*** | 0.32 | 0.90 |
| **Peron's tree frog** | Mann-Whitney U | 618.50 | 4.50 | 42.50 |
| | Z | -0.69 | -0.21 | -1.20 |
| | Asymp. Sig. (2-tailed) | 0.49 | 0.83 | 0.23 |
| **Spotted marsh frog** | Mann-Whitney U | 511.50 | 4.00 | 101.00 |
| | Z | -4.30 | -0.24 | -0.72 |
| | Asymp. Sig. (2-tailed) | **0.00*** | 0.81 | 0.47 |
| **Plains Froglet** | Mann-Whitney U | 62.50 | not present | 34.50 |
| | Z | -1.73 | | -0.88 |
| | Asymp. Sig. (2-tailed) | 0.08 | | 0.38 |

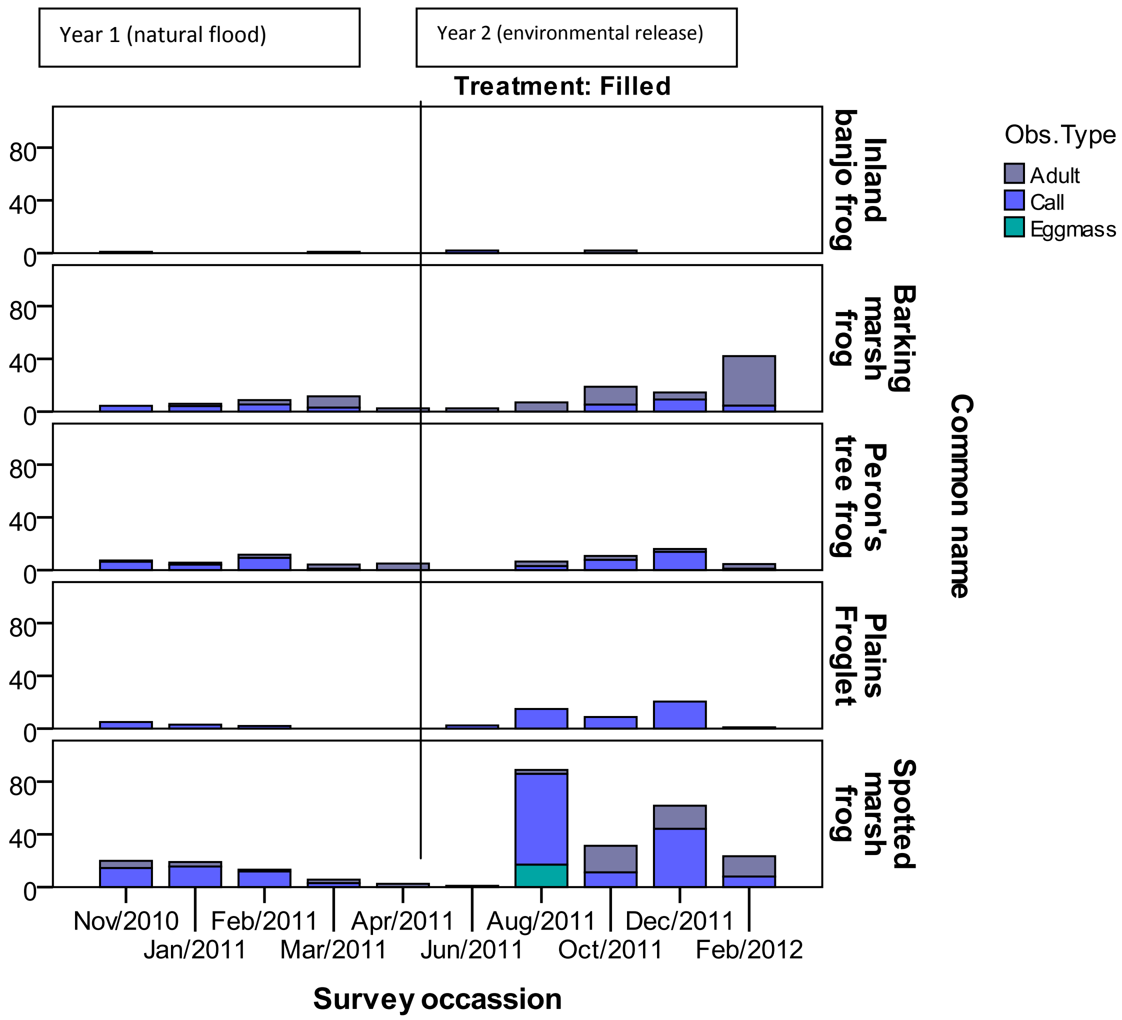



Figure 16. Change in the abundance of frogs (adult and calling) and egg masses between November 2010 and February 2012 (Nov 2010-Apr-2011 from Wassens and Amos, (2011). Tadpoles have been excluded due to different survey methods being used between years.

### *Tadpole development*

The spotted marsh frog/barking marsh frog tadpoles were by far the most abundant tadpole recorded (67 tadpoles) (Figure 17) compared with just six Peron's tree frog tadpoles and one Plains froglet tadpole. Similar results have been obtained in other regions including the Lower Murrumbidgee Floodplain (Spencer and Wassens 2009).

Tadpole abundance and the ranges of development stages changed over time. No tadpoles were recorded during June 2011, by August 2011 egg masses were present but no tadpoles were recorded which suggests that breeding had only just commenced during that survey period. By October 2011, tadpoles were more abundant representing a range of development stages ranging from 30 (recently hatched) to stage 40 (hind limb buds developed). Fewer early stage tadpoles were recorded in December 2011 although a small number of individuals from stages 28 to 40 were present. Only two tadpoles were recorded in February 2012 and both were at later development stages. This indicates that breeding activity had slowed by December 2011 and was complete by February 2012 within the wetlands with the onset of wetland draw down.

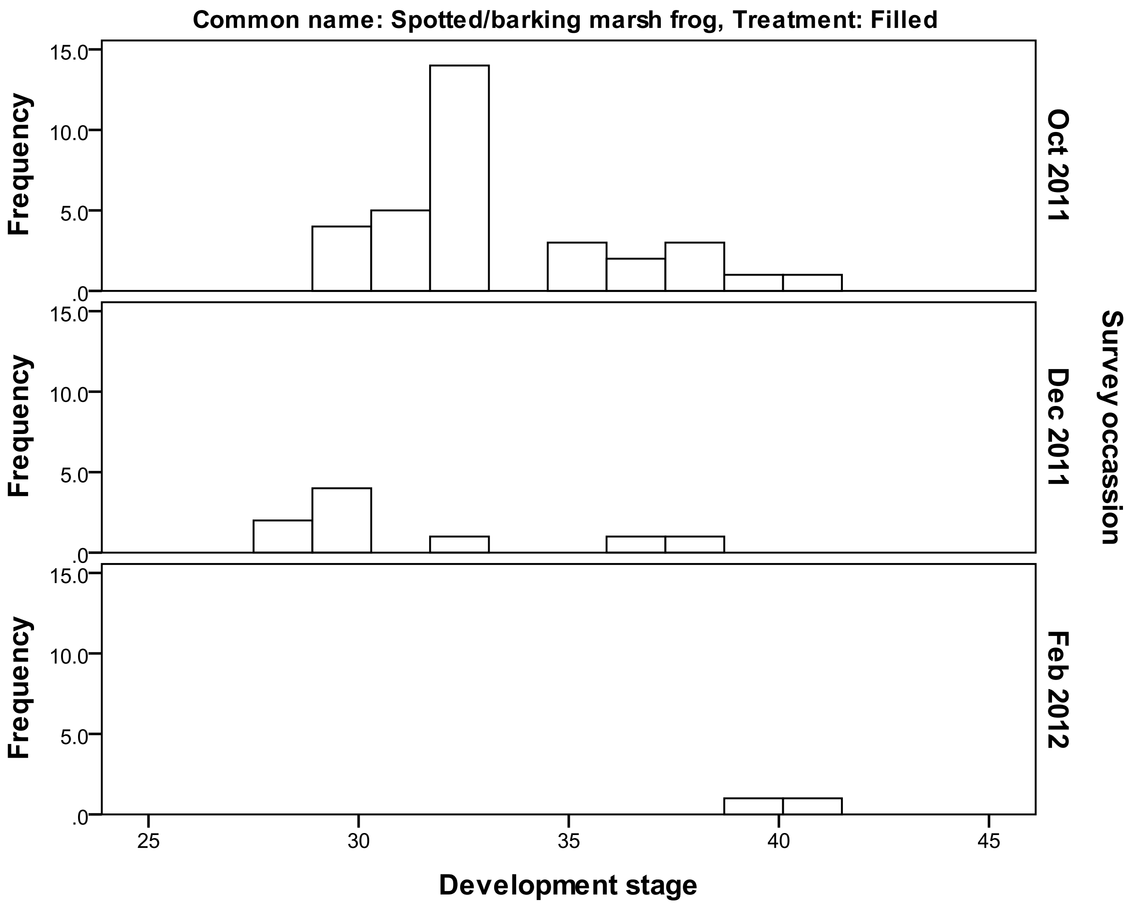



Figure 17. Observation frequencies of the most commonly observed tadpole, the spotted marsh frog/barking marsh frog tadpoles. Development frequencies are not shown for the control wetlands due to small number of tadpoles recorded.

# Fish

Fish communities were surveyed on four occasions from August 2011 to February 2012, at a subset of the 12 wetlands (Euroley (control 1), Molleys, Dry Lake, Yarrada, Sunshower, Gooragool and Mckennas (filled) using a combination of large and small fyke nets, bait traps and sweep netting. Seine netting was used on occasions when the wetland was too shallow to set nets. All fish collected in nets were counted and the first 50 of each species within each net were measured to provide information on the proportion of adult and juveniles of each species within each site. ANOSIM (Primer version 5) was used to test for significant differences in fish community composition over time.

## Key trends

-Five native species were recorded within filled wetlands, three more species than recorded for the Mid-Murrumbidgee during large scale surveys in 2005.

-Native species dominated filled wetlands (were more abundant) than introduced species

- Juveniles of all five native species were recorded in filled wetlands, with juveniles of carp gudgeon, bony bream and un-specked hardyhead making up 50% of more of the total catch for these species by February 2012.

-Five species of introduced fish were recorded, including Oriental weatherloach which has not been previously recorded in the mid-Murrumbidgee.

-Fish communities changed significantly over time, this was partly due to the increasing abundance of juvenile fish following breeding within the wetlands, subsequent reconnection of the wetlands in September 2011 (natural event) and at a subset of wetlands in December 2011 (environmental release) may have also contributed to these differences because they allowed species that only disperse in spring and summer to recolonise the wetlands.

### 3.1.3. Fish

Five species of native fish (carp gudgeon, Australian smelt, un-specked hardyheads, Murray-Darling rainbow fish (Plate 8) and bony bream) were recorded within the filled wetlands (Table 4). Two native fish species were recorded in Euroley (Control 1) (carp gudgeon and Australian smelt). Five species of introduced fish were recorded, with carp and gambusia the most commonly recorded. Native invertebrates - freshwater shrimp *Macrobrachium* spp. and yabbys *Cherax destructor* were also widespread within filled wetlands. Overall, native species were numerically dominant at all filled sites with the exception of Dry Lake (Figure 18). Carp gudgeon was the most abundant and widespread native fish species.

Table 4. Summary of fish species recorded between August 2011 and February 2012

| | Native | | | | | Introduced | | | | | Native Invertebrates | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | Australian Smelt | Bony Bream | Carp Gudgeon | Murray-Darling rainbow fish | Un-specked Hardyhead | Carp | Gambusia | Goldfish | Redfin perch | Weatherloach | Shrimp | Yabby |
| Dry Lake | * | | * | | | * | * | * | * | * | * | * |
| Molleys | * | | * | * | * | * | * | * | * | * | * | * |
| Gooragool | * | * | * | * | * | * | * | * | | | * | |
| Sunshower | * | | * | * | * | * | * | * | * | | * | |
| Yarrada | * | * | * | * | * | * | * | * | * | | * | * |
| Mckennas | * | * | * | * | * | * | * | * | * | | * | * |
| Euroley (C1) | * | | * | | | * | * | * | * | | * | * |
| Berry Jerry [s] | * | | * | | | * | * | | | | * | |
| Coonacoocabil [s] | | | * | | | * | * | | | | * | |
| Narrandera SF [s] | | | * | | | * | * | | | | * | |
| Turkey Flats (C2) [s] | | | | | | | * | | | | | |
| Yanco Ag (C2) [s] | | | | | | * | | | | | | |

[S] sweep net surveys only

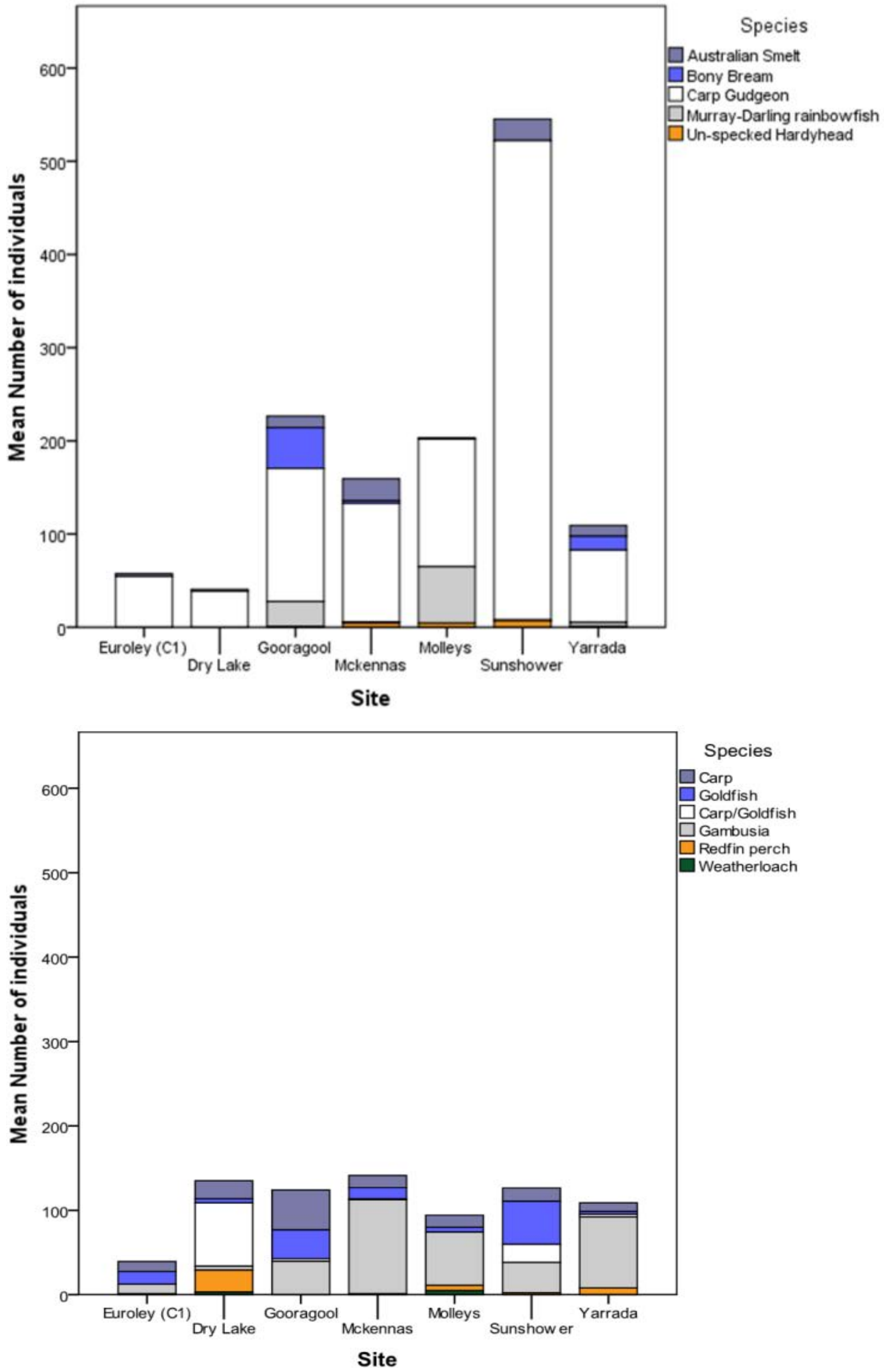



Figure 18. Mean number of individuals of each fish species observed at each wetland: native species (top), exotic species (bottom) (August 2011 – February 2012). Some small carp and goldfish (<25 mm) were combined due to difficulties in separating juveniles of these species in the field.

***Breeding and size structure***

Juveniles of all species were recorded, with the relative proportion of adults to juveniles changing over time (Figure 19). Of the native fish species, carp gudgeon, bony bream and un-specked hardyhead catches represented more than 50% juveniles by February 2012. Juveniles of Australian smelt were present in October and December 2011 which reflects the earlier spawning time for this species. Juveniles of Murray-Darling rainbow fish were present in February 2012 but made up a smaller percentage of the overall catch.

Of the introduced species, juveniles of carp and gambusia made up a similar proportion of the catch for these species on all four survey occasions. In contrast, juvenile goldfish made up a higher percentage of that catch in December 2011 and February 2012. Redfin perch *Perca fluviatilis* (99 individuals) and Oriental weatherloach *Misgurnus anguillicaudatus* (38 individuals) were relatively uncommon throughout the study. A small number of juvenile weatherloach were recorded in August 2011, suggesting that they were young from the previous year, while redfin perch juveniles increased over time making up 100% of the redfin perch catch by February 2012.

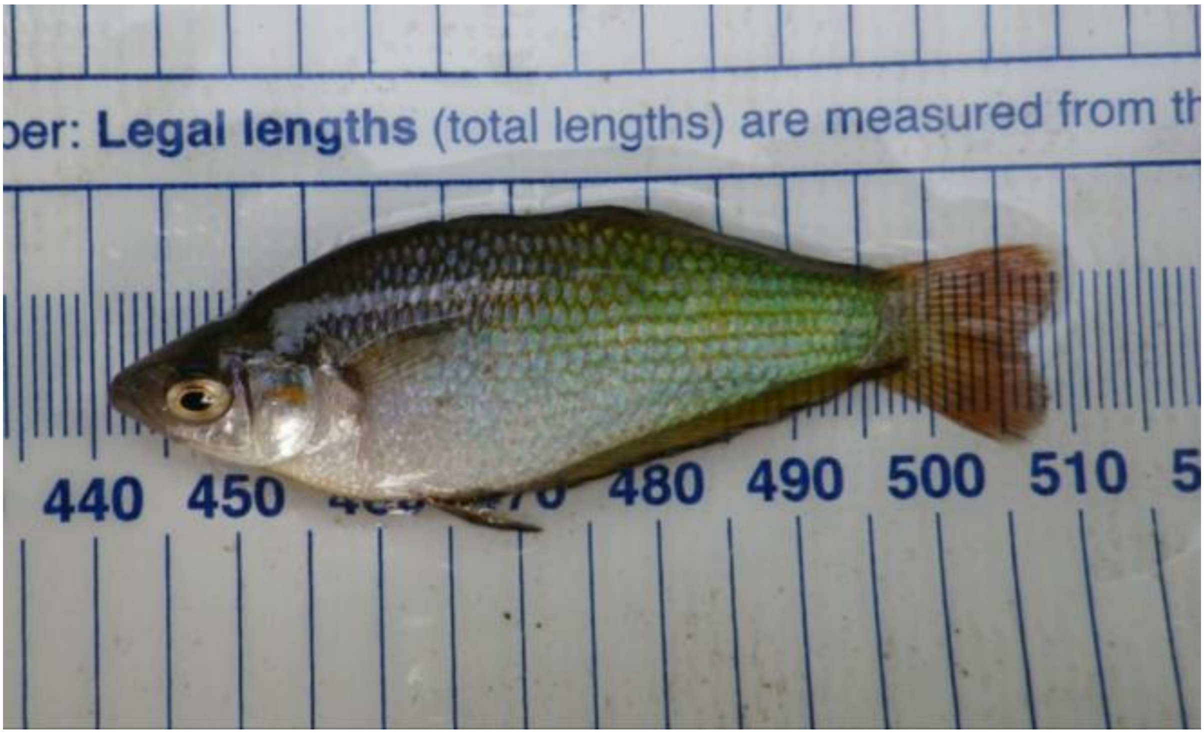



Plate 8. Juveniles of Murray-Darling rainbow fish (adult pictured) were recorded in February 2012.

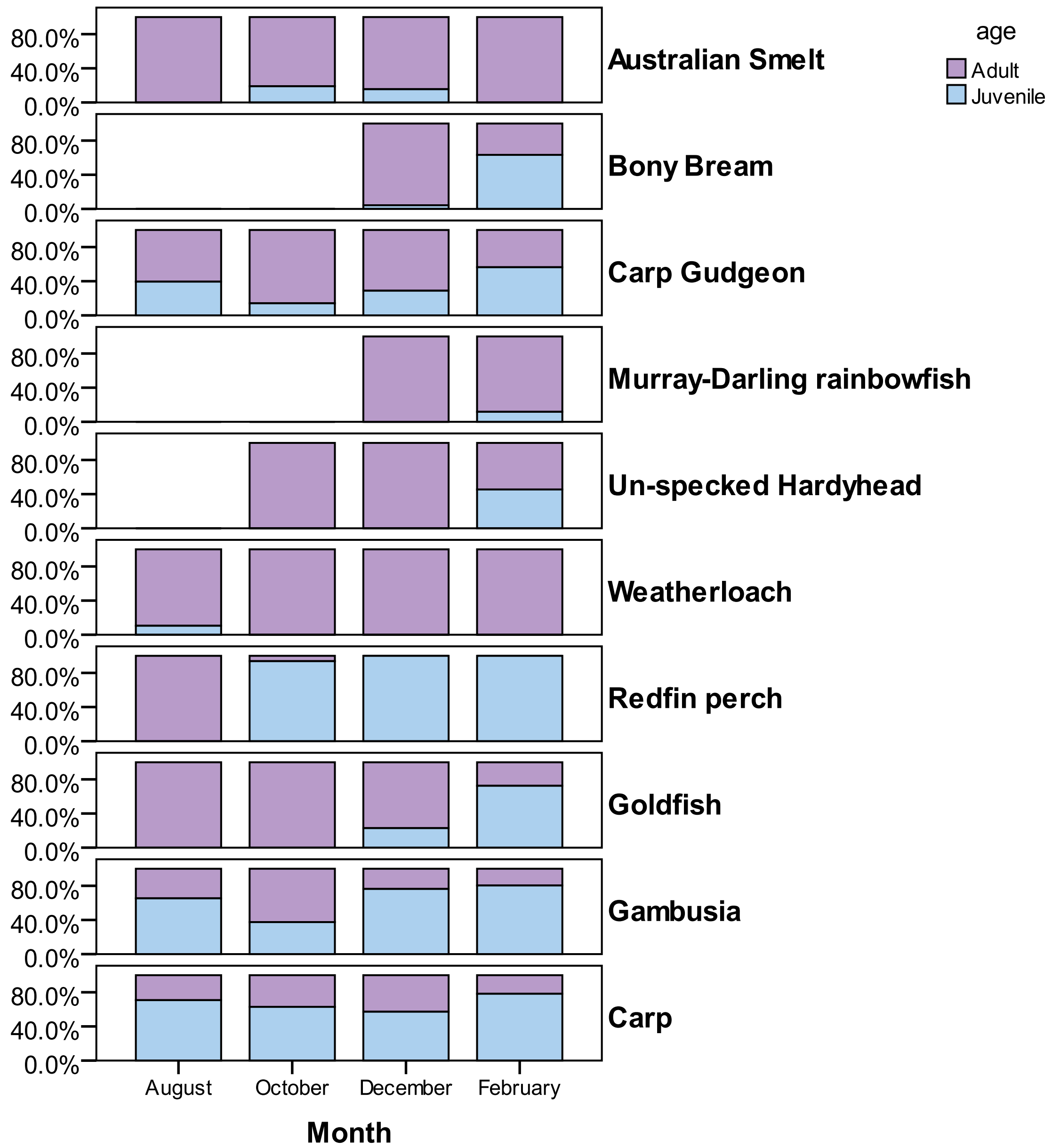



Figure 19. Proportions of adults and juveniles observed for each fish species grouped by survey month.

***Fish communities over time***

Fish communities changed over time (ANOSIM Global R: 0.39, $p < 0.001$). Three native species (un-specked hardyheads, Murray-Darling rainbow fish and bony bream) were not recorded during surveys in August 2011 but were likely to have recolonised wetlands during subsequent top-up flows in September 2011 (natural event) and December 2011 (environmental release). Similar changes in fish communities have been observed in wetlands in the lower Murrumbidgee that were filled in winter with subsequent spring top-up flows and may be explained by the tendency of many native species to move only during warmer temperatures (Spencer and Wassens 2010). This may also explain the absence of these three species from control 1 wetland Euroley. In addition to movement into wetlands, successful spawning and the appearance of juvenile fish within the population also contributed to the shifts in community composition.

# Waterbirds

Ground surveys for waterbirds were carried out in the mid-Murrumbidgee wetlands in June, July, August, October and December 2011, and February 2012. Total counts for each waterbird species and any evidence of breeding activity (nesting and/or the presence of young) were recorded during each survey.

## Key trends

-Waterbird communities were diverse in the filled mid-Murrumbidgee wetlands with 36 species recorded during the 2011-12 surveys.

-Species richness and abundance differed significantly among wetlands and wetlands with high waterbird abundance tending to have most diverse waterbird communities.

-Total numbers of waterbirds varied considerably over time in response to filling during winter and summer months, and the drying-down of many wetlands by December 2011 – February 2012.

-Breeding was detected in eight waterbird species. Overall, colonial waterbird breeding was limited with only small numbers of nests recorded in filled wetlands -Gooragool, Yarrada and McKennas lagoons.

### 3.1.4. Waterbirds

***Species diversity and abundance***

The mid-Murrumbidgee wetlands supported on average 1,040 waterbirds, comprising 36 species, during the 2011-12 surveys. Two species listed on international bird agreements were observed and breeding activity was detected in eight species (Appendix 4). None of the observed species are listed on the NSW *Threatened Species Conservation (TSC) Act 1995* or the Commonwealth *EPBC 1999 Act*.

Species richness and abundance differed significantly among wetlands ($F_{11,55}$ = 2.1, $P$ = 0.03 and $F_{11,55}$ = 2.2, $P$ = 0.03, respectively). Wetlands with high waterbird abundance frequently had high species richness, for example Gooragool and Sunshower lagoons supported high waterbird abundance (Figure 20, Appendix 3) and high species richness Gooragool (24 species) and Sunshower (22 species). Species richness was also high at Dry Lake (22 species).

Australian wood duck *Chenonetta jubata*, grey teal *Anas gracilis*, Pacific black duck *Anas superciliosa* and white-faced heron *Ardea novaehollandiae* were the most widespread species, occurring in every survey site (Appendix 3). We also observed pairs of white-bellied sea eagles *Haliaeetus leucogaster* (listed under the China-Australia Migratory Bird Agreement) at Yarrada, Dry Lake, McKennas and Gooragool lagoons.

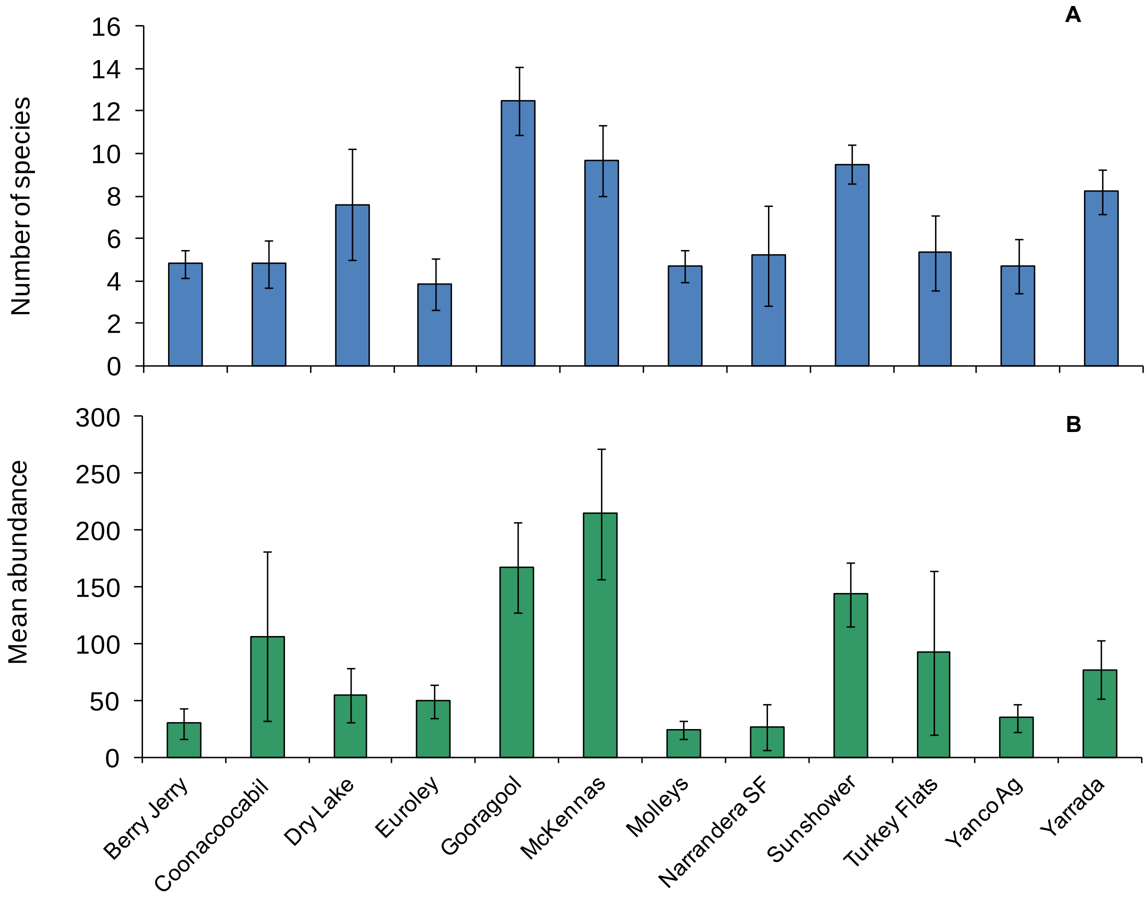



Figure 20. Mean (+/- standard error) (A) number of species and (B) abundance of waterbirds in 12 surveyed wetlands in the mid-Murrumbidgee (June 2011 – February 2012).

Total numbers of waterbirds varied considerably over time in response to inflows during winter and summer months, and the drying-down of many wetlands between December 2011 and February 2012 (Figure 21). The control sites and many sites that were filled from June – August 2011 had dried down by the December 2011 and February 2012 surveys, and fewer birds were recorded. Four of the filled sites received additional inflows in summer (November - December 2011) (Gooragool, Yarrada, Sunshower and Molleys lagoons) and so these sites continued to provide waterbird habitat over summer months (Figure 21). Overall sites that were filled in winter and summer 2011 supported a significantly greater number of species and total abundance of waterbirds compared to the control sites ($F_{2,64} = 6.1$, $P = 0.004$ and $F_{2,64} = 3.6$, $P = 0.033$, respectively).

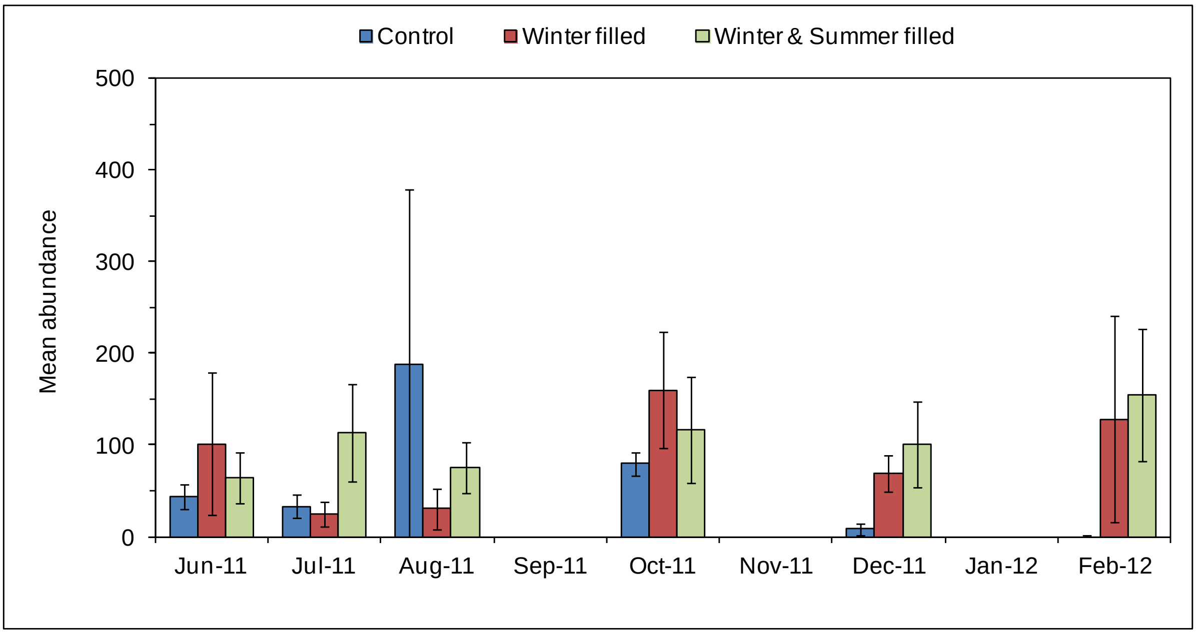



Figure 21. Mean number of waterbirds observed in control sites (n = 3), wetlands filled in winter 2011 (n = 5), and wetlands filled in winter and summer 2011 (n = 4).

Overall waterbird assemblages differed significantly among the types of wetlands (Global R = 0.1, $P$ = 0.027). The relative abundance of each waterbird functional group varied among the wetlands (Figure 22), with dabbling ducks and grazing waterfowl, grey teal, Pacific black duck and wood duck, dominating waterbird assemblages. Shorebirds (small waders) were recorded in Dry Lake and Yarrada lagoon towards the end of 2011 when water levels dropped and areas of mudflat were exposed.

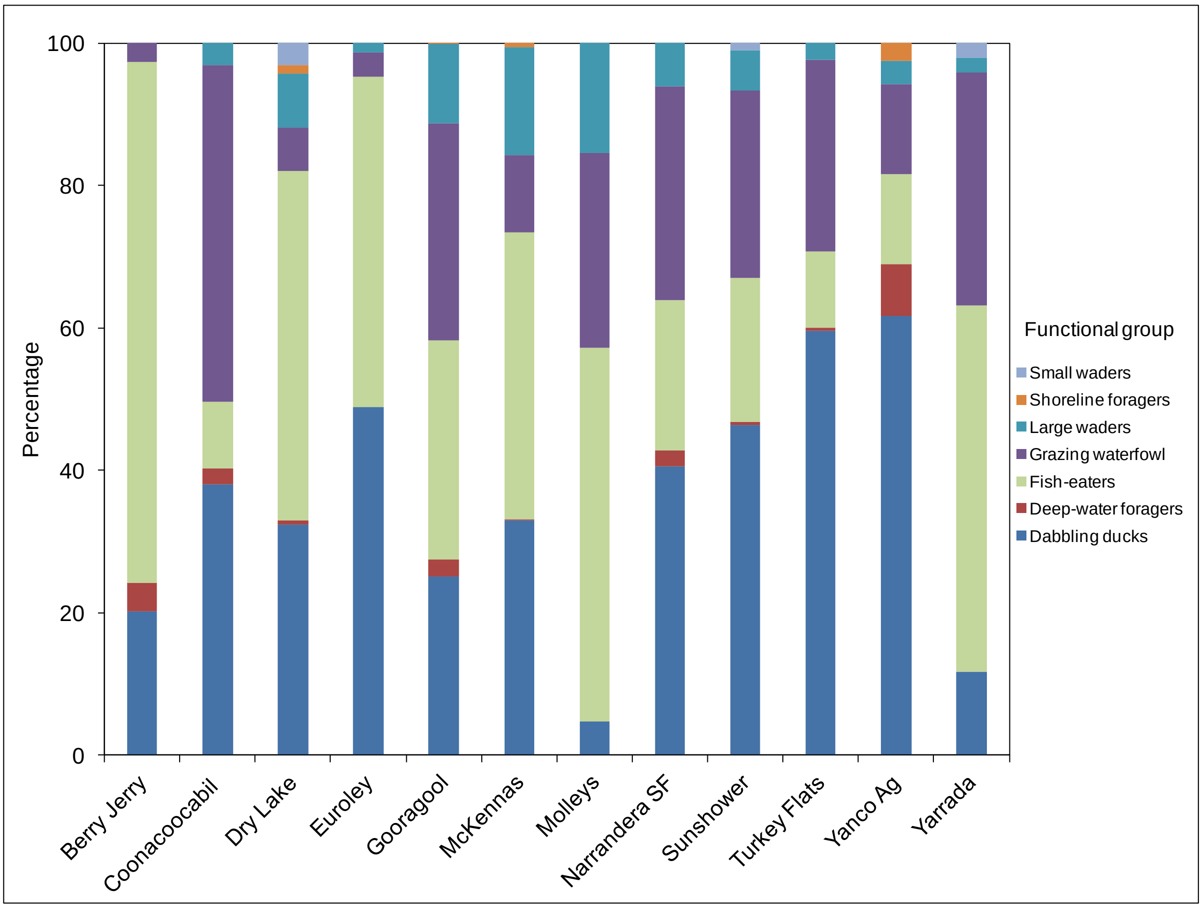



Figure 22. Proportion of seven waterbird functional groups (as per Roshier et al. 2002) in the 12 wetlands surveyed in the mid-Murrumbidgee (June 2011-February 2012).

### *Waterbird breeding*

Six of the surveyed wetlands supported small numbers of breeding waterbirds (Appendix 3). Remains of cormorant and darter nests from the 2010-11 season were observed in small river red gum saplings in Gooragool, McKennas and Yarrada lagoons. Great cormorants *Phalacrocorax carbo* and darters *Anhinga melanogaster* also nested in these sites in small numbers during the 2011-12 season. Most activity was observed in Yarrada lagoon where approximately 44 active nests of darters (23 nests), great cormorants (16 nests) and little pied cormorants *Phalacrocorax melanoleucos* (5 nests) were observed (Plate 9). Nesting at Yarrada was first observed in October 2011 and birds nested through to February 2012. Most nesting was complete by February 2012 with only two darter and one great cormorant nest with fledglings observed in Gooragool lagoon during surveys. Two darter nests were also observed in McKennas lagoon during the December surveys.


(a)


(b)

Plate 9. Small numbers of cormorants and (a) darters nested in (b) Yarrada lagoon in 2011-12.

### 3.1.5. Freshwater turtles

We caught both Macquarie River *Emydura macquarii* and long-necked *Chelodina longicollis* turtles (11 in total) in three of the wetlands surveyed between August 2011 and February 2012. Three adult long-necked turtles were caught in total, one in Gooragool lagoon and two in Molleys lagoon. Macquarie River turtles were detected in Yarrada (5 turtles) and Molleys (3 turtles), with recently hatched Macquarie River turtles detected in both of these sites in the December 2011 and February 2012 surveys (Plate 10).

Plate 10. Hatchling Macquarie River turtle (*Emydura macquarii*) from Yarrada lagoon.

# Biofilms and macroinvertebrates

Monitoring of in-stream parameters (biofilms and macroinvertebrates) was undertaken at three reaches in the Murrumbidgee River downstream of Burrinjuck Dam and four reaches in the Tumut River downstream of Blowering Dam. Sampling was also undertaken in the Goobarragandra River, an unregulated tributary of the Tumut River that will serve as a reference site for this study. Sampling was undertaken on five occasions between June and October 2011; once prior to the first flow pulse and four times after the environmental flow.

## Key trends

-The environmental flow significantly reduced the biomass of biofilm, most likely due to scouring of biofilms from increased water velocity. Biofilm biomass returned to their previous state more quickly within the Tumut compared to the Murrumbidgee River

-The composition of biofilms changed following the environmental release, in the Murrumbidgee River the environmental flow reduced the relative proportion of red, green and blue-green algae and increased diatoms- these positive changes were maintained within the Murrumbidgee for an extended period after the release. In the Tumut River the environmental flow initially reduced green algae's but these quickly returned to their previous levels when dam releases returned to their normal regulated levels.

-For majority of the dominant macroinvertebrate taxa there was a short term increase in abundances immediately after the environmental release, followed by a trend of abundances returning to a level similar to that observed prior to the environmental flow.

## 3.2. Monitoring of in-stream parameters

### 3.2.1. Biofilm organic biomass

A comparison of biofilm organic biomass prior to the environmental flow (3/6/2011) to that immediately after the environmental flow (28/6/2011) demonstrates the immediate response of biofilm to the environmental flow event. The environmental flow significantly reduced the biomass of biofilm at Murrumbidgee sites 1 and 2 and Tumut sites 2 and 3, most likely due to scouring of biofilms from increased water velocity (Figure 23).

By the third sample date on 30$^{th}$ July (approximately 5 weeks after the recession of the environmental flow) the biomass of biofilm at most sites had significantly increased to levels higher than levels observed prior to the environmental flow, with the exception of the downstream sites in the Tumut River (Tumut 3 and 4) (Figure 23). By September the biomass of biofilm was very high at several sites, particularly at Tumut site 1, relative to the unregulated reference reach in the Goobarragandra River where biofilm biomass remained consistently low throughout the study (Figure 23).

A reduction in biofilm biomass was observed at Murrumbidgee site 1 between the 3$^{rd}$ and 4$^{th}$ sample date when the biomass Murrumbidgee site 2 was increasing and Murrumbidgee site 3 was relatively constant. The biofilm at site 1 was most likely scoured by a large flow on 4$^{th}$ September. This site could not be sampled on 4$^{th}$ September when all other sites were sampled, because it was unsafe to sample the reach, but was instead sampled on 6$^{th}$ September after the flow. If this site had been sampled prior to the release the site would most likely have followed a similar trajectory of increasing biofilm biomass similar to that observed at Murrumbidgee site 2 (Figure 23). The biofilm chlorophyll-*a* levels showed a similar pattern of response to the organic biomass (Figure 24). Chlorophyll-*a* levels were extremely high relative to the unregulated Goobarragandra River.

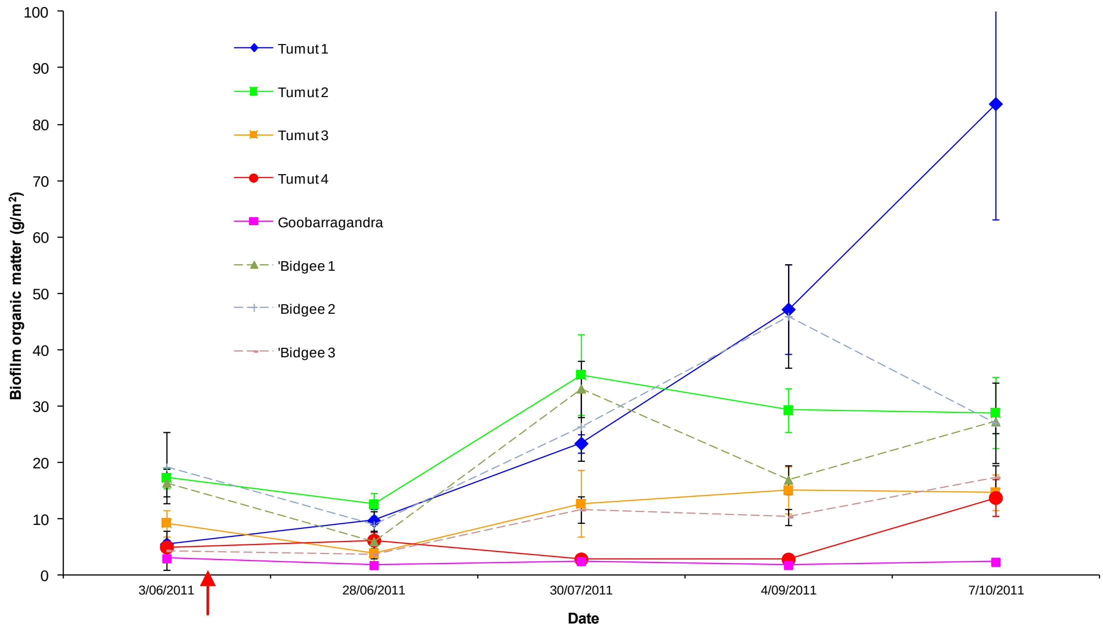



Figure 23. Biofilm organic biomass (dry weight mg/m$^2$) on cobbles at reaches in the Murrumbidgee River, Tumut River and Goobarragandra River. Red arrow indicates the time of peak discharge for the environmental flow pulse.

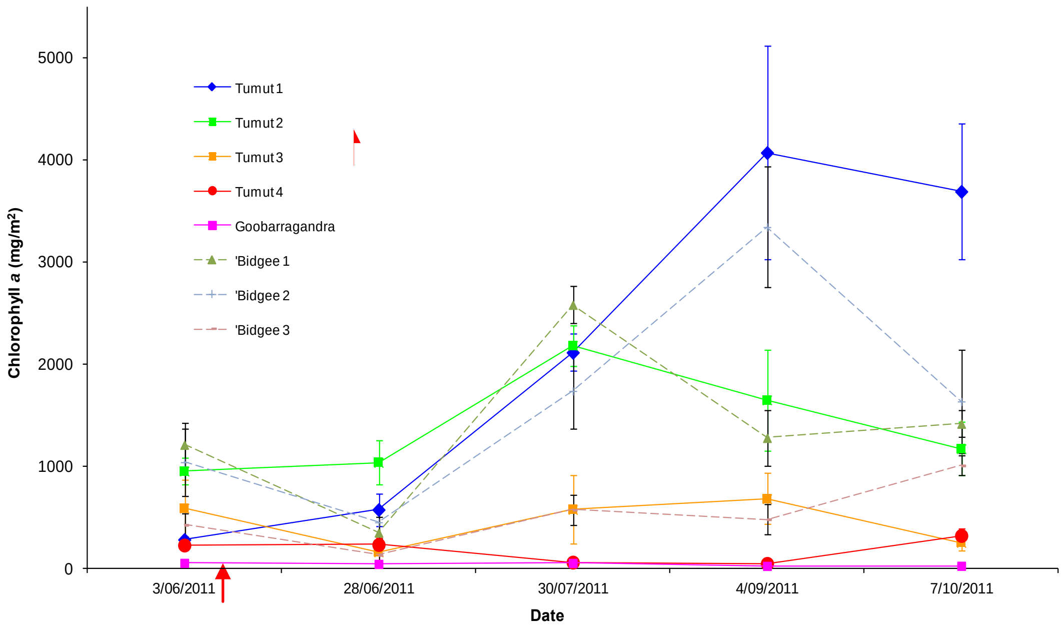



Figure 24. Biofilm Chlorophyll-*a* (mg/m$^2$) on cobbles at reaches in the Murrumbidgee River, Tumut River and Goobarragandra River. Red arrow indicates the time of peak discharge for the environmental flow pulse.

### 3.2.2. Biofilm composition

A total of 58 algal taxa were identified during the study. This included one red alga (Rhodophtya), ten green algae (Chlorophyta), nine blue-green algae (Cyanophyta) and 38 diatoms (Bacillariophyta) (Appendix 6). The common taxa were Audonella (red algae) (Plate 11a), Draparnaldia (green alga) (Plate 11b) and *Cymbella cistula* (diatom) (Plate 11c) and *Melosira varians* (diatom) (Plate 11d) (Table 5). The cosmopolitan diatom *Melosira varians* was common (5 - 10%) throughout the study at all sites.

(a)  (b) 

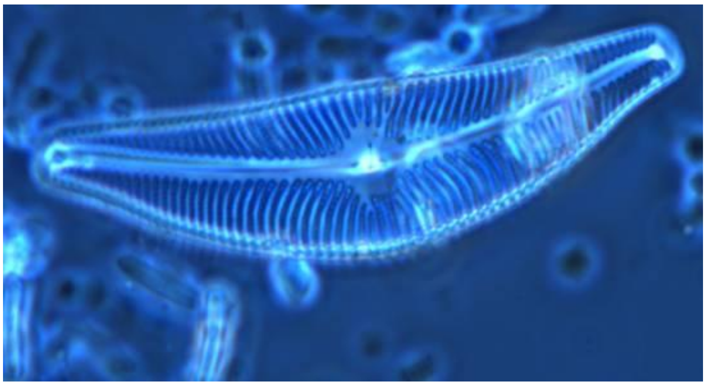
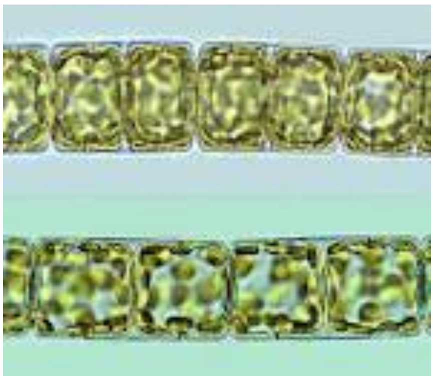
(c)  (d) 

Plate 11. Common in-stream biofilm taxa (a) Red Alga – Audouinella, (b) Green alga – Draparnaldia, (c) Diatom – *Cymbella cistula,* (d) Diatom – *Melosira varians* (Photo: www.diatomloir.eu/Diatodouces/Chloroplastes.html)

Prior to the environmental flow, green algae in the Murrumbidgee River were 16% of the total biovolume at Murrumbidgee site 1 and less than 1% of the total biovolume at Murrumbidgee sites 2 and 3. Tumut River sites 1, 2, and 3 all had greater than 20% biovolume of green algae prior to the environmental flow (there were no data for Tumut site 4 prior to the environmental flow) (Figure 25 and Figure 26). In contrast, green algae were absent from the Goobarragandra River (reference stream) prior to the environmental flow and throughout the study and the diatom assemblage was mixed, with no single species dominating the biofilm (Table 5).

The environmental flow significantly altered the relative abundance of algal divisions in the Murrumbidgee and Tumut Rivers (Figure 25 and Figure 26). In the Murrumbidgee River, the environmental flow reduced the relative proportion of red, green and blue-green algae when compared to samples collected prior to the environmental flow. By 30/7/2011 (sample date 3) all sites were dominated by diatoms (equal to or greater than 93% of total biovolume) (Figure 25). At Murrumbidgee site 1 the diatom *Melosira varians* ranged from 22-46% as part of a mixed diatom assemblage. The filamentous green alga *Oedogonium* returned to this site in October (Table 5). The Murrumbidgee site 2 was initially a mixed community of 20% red algae and 65% diatoms which increased to a diatom dominance of 95% by 4/8/2011. There was almost a mono culture of *Cymbella cistula* (92%) in mucilaginous sheaths in September at this site but reduced to 28% in October with a shift to a mix of the diatoms *Melosira* and *Navicula* (Table 5). Murrumbidgee site 3 was initially a mix of 50% red *Audounella* and 30% diatom *Melosira* . After the environmental flow there was a shift to 95% diatoms (Figure 25).

In the Tumut River the environmental flow reduced the relative biovolume of green algae at Tumut sites 1, 2 and 3 from approximately 20% at all sites down to nil at sites 1 and 3 and 11% at site 2 by 28/6/2011 (Figure 26). However, the effect of the environmental flow on biofilms in the two most upstream sites in the Tumut River was relatively short lived. By the 30/7/2011 the proportion of green algae increased significantly at Tumut sites 1 and 2 and it increased further by 4/9/2011. At Tumut site 1 the red algae *Audounella* was totally removed by the environmental flow and replaced with the branching green algae *Draparnaldia*, reaching 94% of biofilm on 4/9/2011 (Table 5). This increase in green algae was not observed at Tumut sites 3, and 4 which are sites downstream of the confluence with the unregulated Goobarragandra River are influenced by the flows in that unregulated river system.

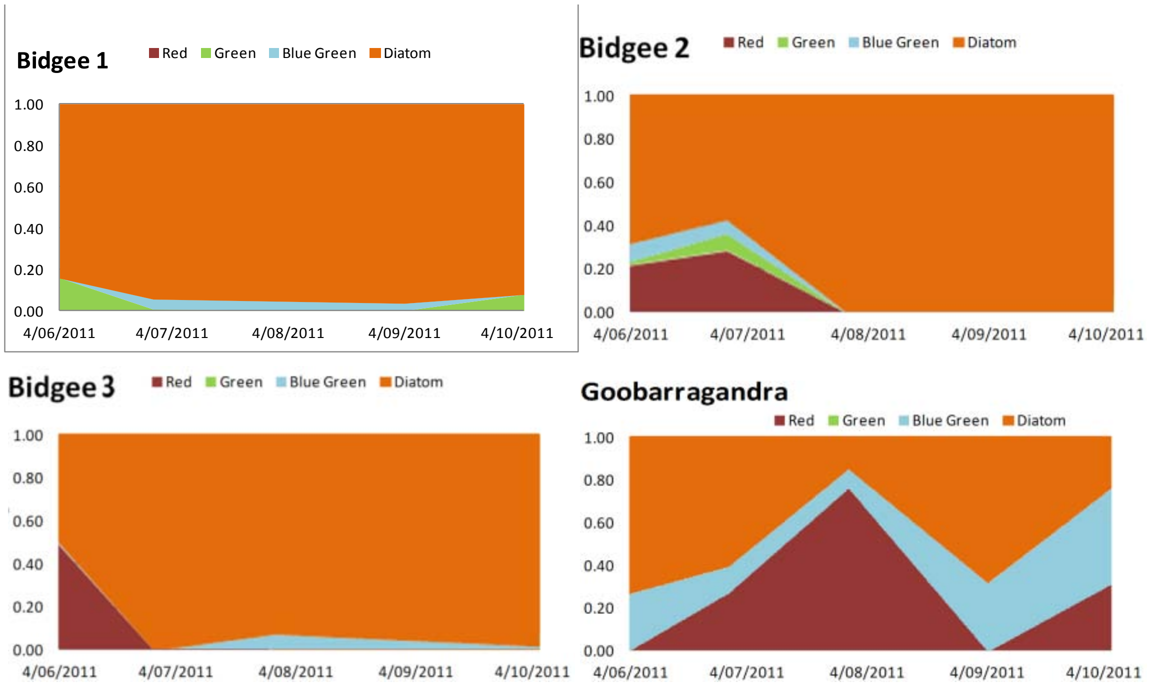



Figure 25. Relative abundance of major algal divisions (red, green, blue-green, diatoms) between June 2011 and October 2011 at three sites in the Murrumbidgee River and the Goobarragandra River (reference).

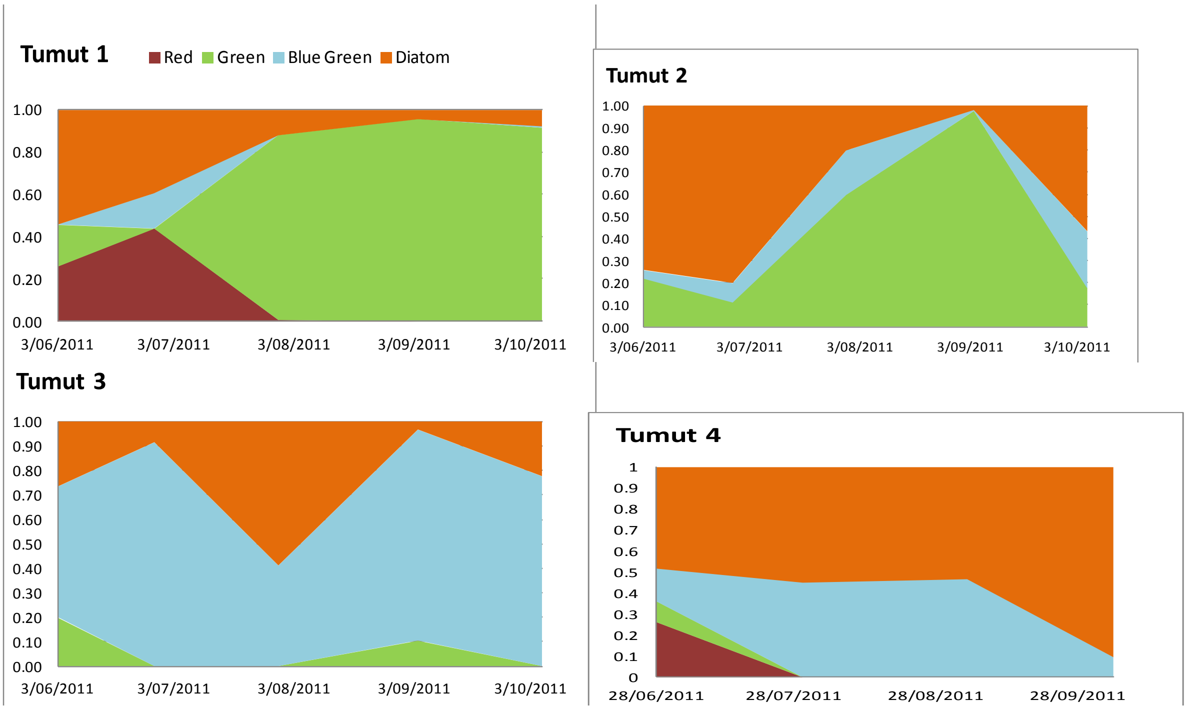



Figure 26. Relative abundance of major algal divisions (red, green, blue-green, diatoms) between June 2011 and October 2011 at four sites in the Tumut River. There was no data for Tumut site 4 on 3/6/2011, so the first sample presented is from 28/6/2011 after the environmental flow.

Table 5. Summary of algal taxa occurring > 10% at any site or sampling time.

| *Site* | *Sample event* | *Audounella* | *Draparnaldia* | *Oedogonium)* | *Leptolyngbya* | *Lyngbya* | *Oscillatoria* | *Phormidium* | *Achnanthes cf.subexiqua* | *Cocconeis placentula* | *Cymbella cistula* | *Cymbella tumidia* | *Encynema silesiacum* | *Encynema minutum* | *Gomphonema parvulum* | *Gomphonema clavatum* | *Melosira varians* | *Navicula gregaria* | *Nitzschia frustulum* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Bidgee1** | 1 | | | **0.15** | | | | | | | | | | | | | **0.46** | | |
| | 2 | | | | | | | | | **0.12** | | | | | | | **0.43** | | |
| | 4 | | | | | | | | | | | | | | | | **0.48** | | |
| | 5 | | | | | | | | | | | | | | | | **0.22** | | |
| **Bidgee2** | 1 | **0.22** | | | | | | | | | | | | | | | **0.17** | **0.41** | |
| | 2 | **0.29** | | | | | | | | | | | | | | | **0.35** | | |
| | 3 | | | | | | | | | | | | | | | | **0.31** | **0.51** | |
| | 4 | | | | | | | | | | **0.92** | | | | | | | | |
| | 5 | | | | | | | | | **0.18** | **0.28** | | | | | | **0.33** | | **0.12** |
| **Bidgee3** | 1 | **0.50** | | | | | | | | | | | | | | | **0.14** | **0.29** | |
| | 2 | | | | | | | | | | | | | | | | **0.65** | | |
| | 3 | | | | | | | | | **0.12** | | | | | | | **0.33** | **0.36** | |
| | 5 | | | | | | | | | | | | | | | | **0.54** | | |
| **Gooba** | 1 | | | | | **0.21** | | | | | | | | | **0.18** | | **0.17** | **0.10** | |
| | 2 | **0.27** | | | | | | | | | | | | | | **0.28** | | | |
| | 3 | **0.76** | | | | | | | **0.12** | | | | | | | | | | |
| | 4 | | | | | | **0.14** | | | | | **0.25** | | | **0.11** | | **0.21** | | |
| | 5 | **0.31** | | | **0.44** | | | | | | | | | | | | **0.12** | | |
| **Tumut1** | 1 | **0.26** | | | | | | | | | | | | | | | **0.37** | | |
| | 2 | **0.44** | | | | | **0.12** | | | | | | | | | | **0.23** | | |
| | 3 | | **0.63** | **0.25** | | | | | | | | | | | | | | | |
| | 4 | | **0.94** | | | | | | | | | | | | | | | | |
| | 5 | | **0.87** | | | | | | | | | | | | | | | | |
| **Tumut2** | 1 | | | **0.10** | | | | | | | | | | | | | **0.54** | | |
| | 2 | | | **0.11** | | | | | | | | | | | | | **0.75** | | |
| | 3 | | **0.60** | | | | | | | | | | | | | | | | |
| | 4 | | **0.96** | | | | | | | | | | | | | | | | |
| | 5 | | | **0.18** | | | | | | | **0.21** | | | | | | | | |
| **Tumut3** | 1 | | **0.20** | | | | | **0.32** | | | | | | | | | **0.13** | | |
| | 2 | | | | | **0.91** | | | | | | | | | | | | | |
| | 3 | | | | **0.15** | | | | **0.17** | | | | | | **0.14** | | | | |
| | 4 | | **0.10** | | | | **0.85** | | | | | | | | | | | | |
| | 5 | | | | | | **0.78** | | | | **0.19** | | | | | | | | |
| **Tumut4** | 2 | **0.26** | | **0.10** | | | | | | | | | | | | | **0.29** | | |
| | 3 | | | | | **0.45** | | | | | | | **0.20** | **0.20** | | | | | |
| | 4 | | | | | **0.37** | | | | | | **0.17** | | | | | **0.24** | | |
| | 5 | | | | | | | | | | **0.42** | | **0.15** | | | | **0.16** | | |

***Aquatic macroinvertebrates***

A total of 64,568 macroinvertebrates were collected during the study. The 10 most dominant taxa, in descending order were Oligochaeta (worms), Corixidae (bugs), Caenidae (mayflies), Orthocladinae (dipteran flies), Baetidae (mayflies), Chironominae (dipteran flies), Gripoptyerigidae (stone flies), Simulidae (dipteran flies), Lephtophleibidae (mayflies) and Hydropsychidae (caddisflies).

There were generally a lower number of macroinvertebrate taxa in the Murrumbidgee and Tumut rivers than in the Goobarragandra River throughout the study, with the exception being that the Murrumbidgee River had a similar number of taxa as the Goobarragandra River prior to the environmental flow (Figure 27). There was a significant increase in the number of macroinvertebrate taxa in the Tumut River immediately after the environmental flow (Figure 27). However, the total number of taxa in the Tumut River significantly reduced by sample 3/9/11 after a sequence of three managed flow releases in this system. There was also a reduction in the number of taxa observed in the Murrumbidgee River.

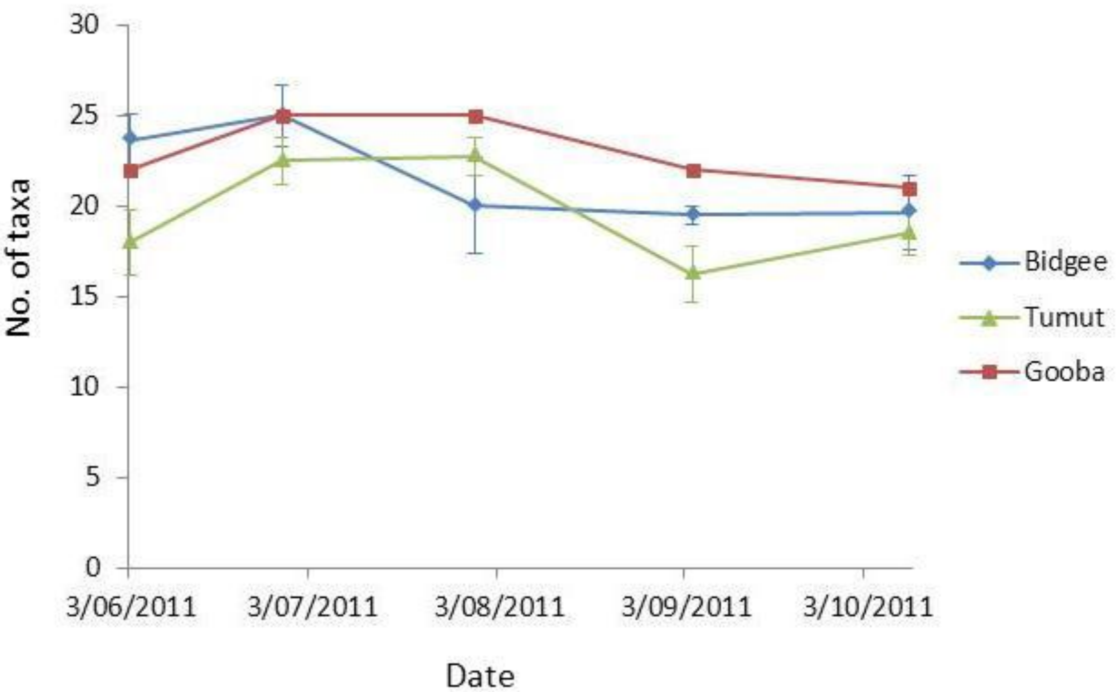



Figure 27. Mean (±1 SE) total number of taxa found in the Murrumbidgee (blue), Tumut (green) and Goobagandra river (red).

The abundance and relative proportions of macroinvertebrate taxa provide a more complete description of the response to the environmental flow. There was an increase in the abundance of macroinvertebrates in both the Murrumbidgee River and Tumut River immediately after the environmental flow (Figure 28). This increase was partly due to an increase in the number of Letptophlebiidae mayflies (Figure 29), Gripoterygidae stoneflies (Figure 29) and Baetidae mayflies (Figure 30).

Caenidae mayflies, Hydropsychidae caddisflies and Chironomid and Simuliidae flies all increased in the Tumut River immediately after the flow, but there were no similar increases in the Murrumbidgee or Goobarragandra Rivers (Figure 31). The Tumut River experienced a high level of disturbance due to the flows cycling from baseflow to almost bankfull several times during the study period. Abundances of Corixids declined in both the Tumut and Murrumbidgee rivers after the environmental flow (Figure 32). This trend was not observed in the reference Goobarragandra River.

The remaining two dominant taxa; Oligochaeta and Orthocladiinae, provided no clear indication of a response to the environmental flows. While abundances remained relatively stable for both taxa in the Goobarragandra River, abundances in the Tumut and Murrumbidgee Rivers varied less predictably over time with considerable variability among samples resulting in large standard errors (Figure 33).

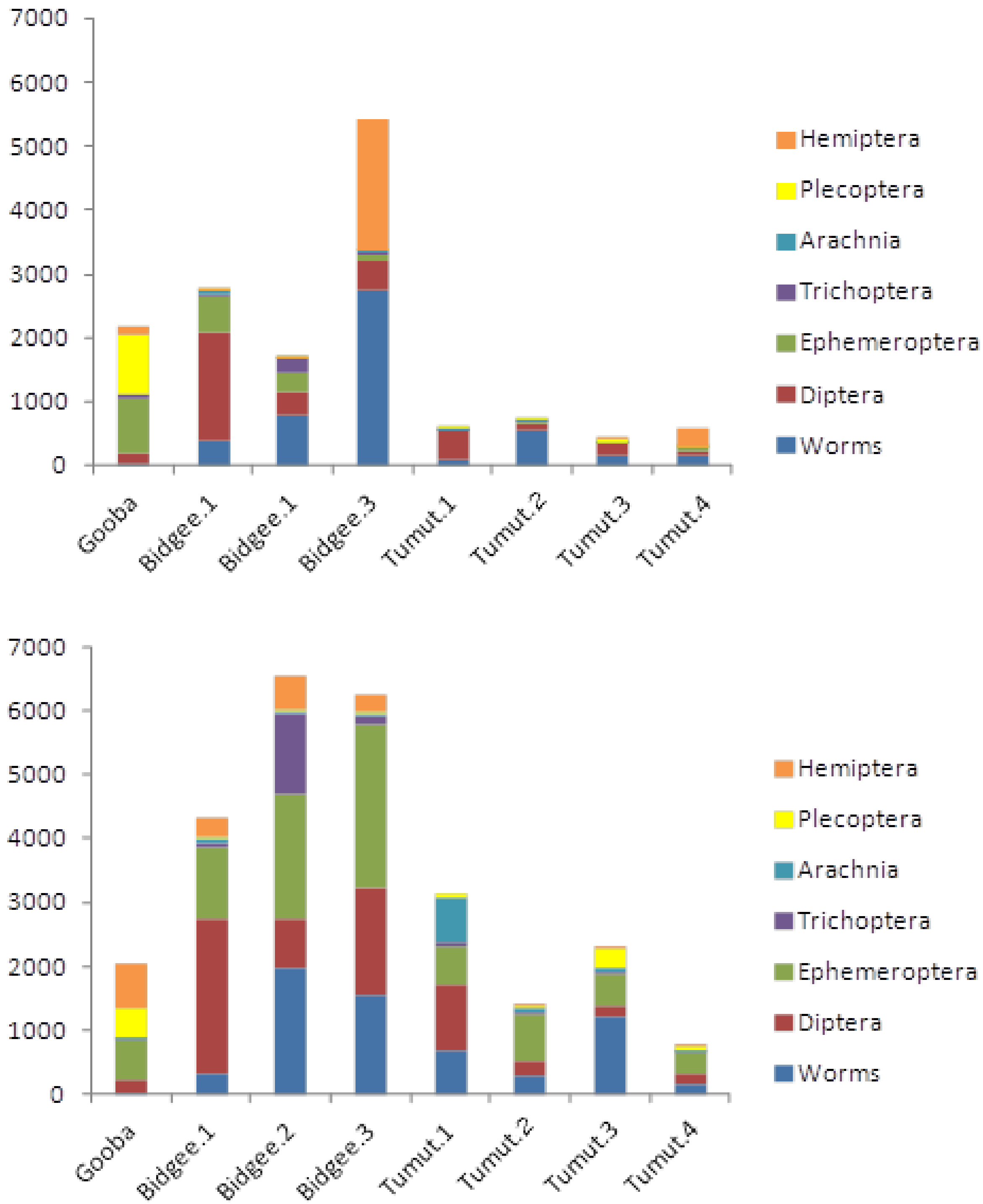



Figure 28. Site-specific total abundances of macroinvertebrates in edge, riffle and pool habitats combined, a) before the environmental flow (3/6/2011) and b) after the environmental flow on 28/6/2011. Relative contribution by the major macroinvertebrate groups is highlighted.

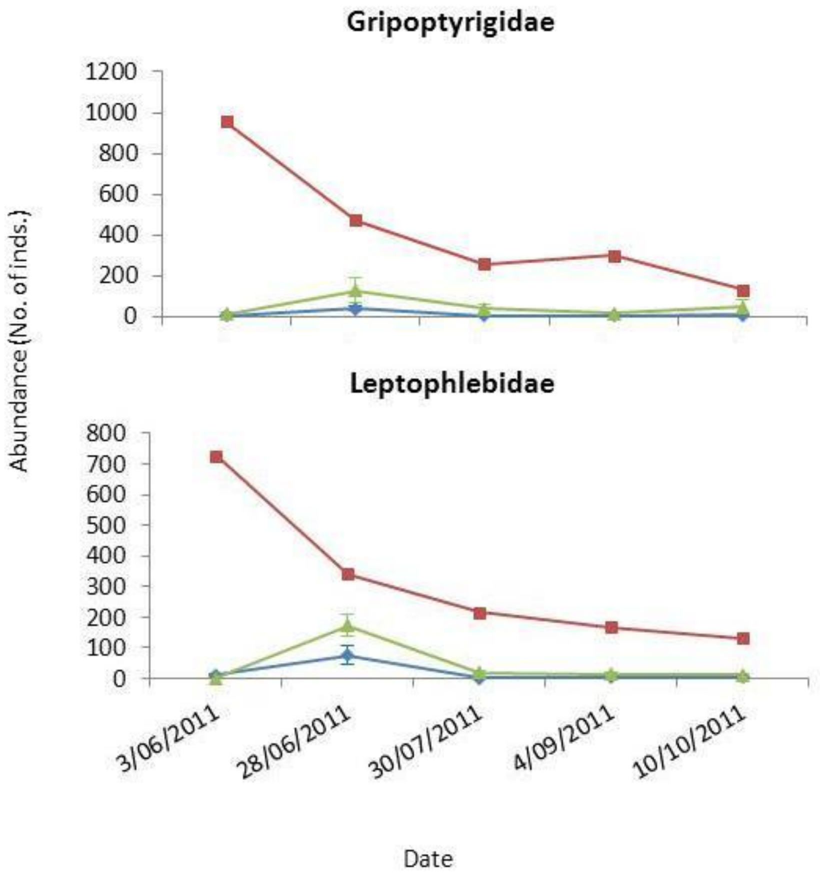



Figure 29. Mean (±1 S.E) abundance of the families Gripoptyrigidae and Leptophlebeidae in the Murrumbidgee, Tumut and Goobarragandra rivers over the study period.

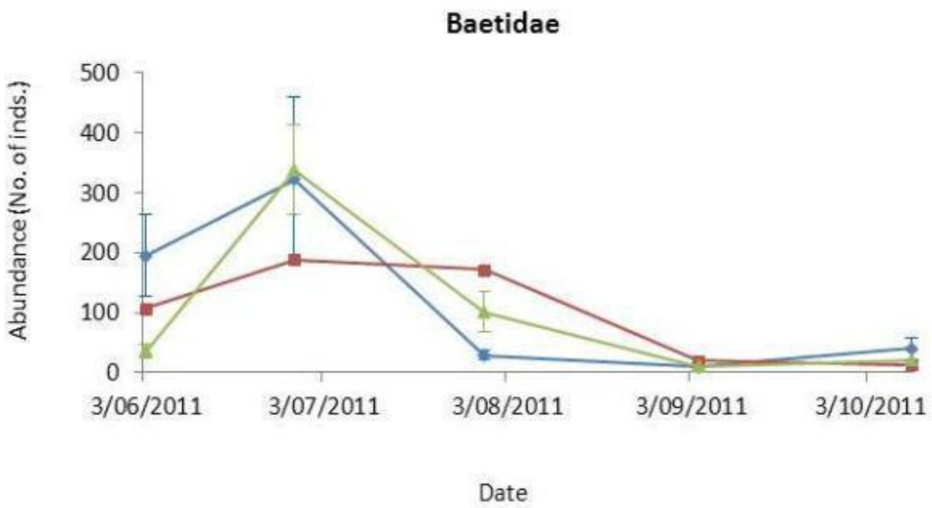



Figure 30. Mean (±1 S.E) abundance of Baetid mayflies in the Murrumbidgee, Tumut and Goobarragandra rivers over the study period.

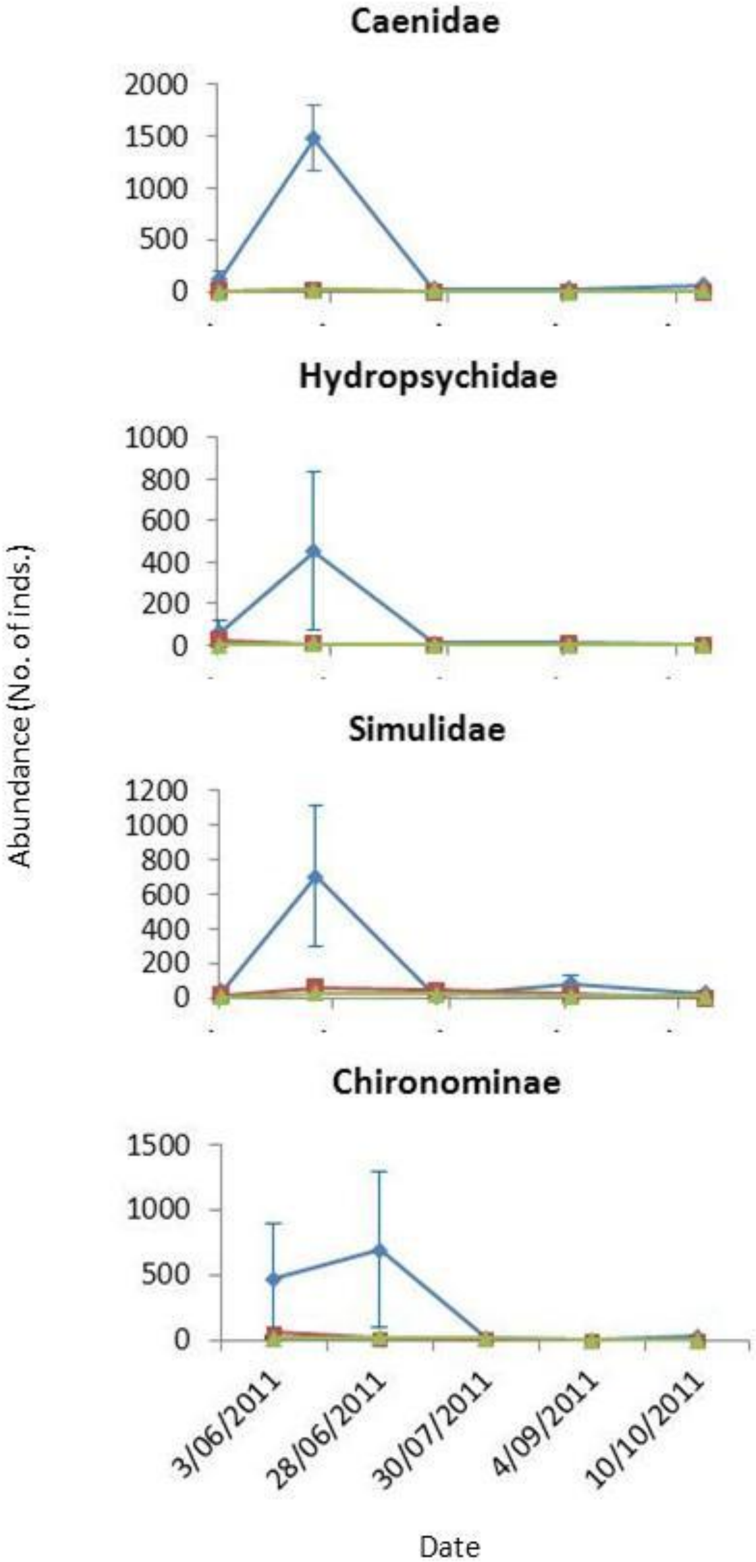



Figure 31. Mean (±1 S.E) abundance of Caenids, Hydropsychids, Simulids and Chironomids in the Murrumbidgee (blue), Tumut (green) and Goobarragandra (red) rivers over the study period.

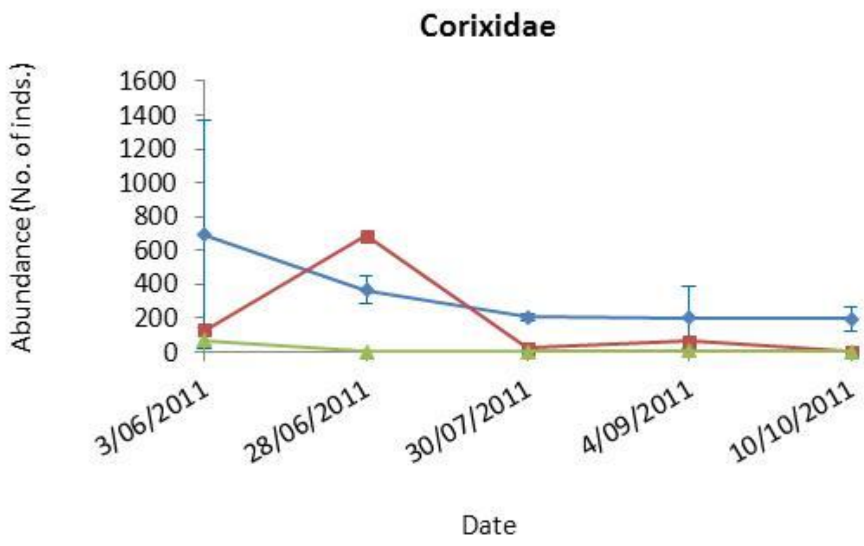



Figure 32. Mean (±1 S.E) abundance of Corixids in the Murrumbidgee (blue), Tumut (green) and Goobagandra rivers (red) over the study period.

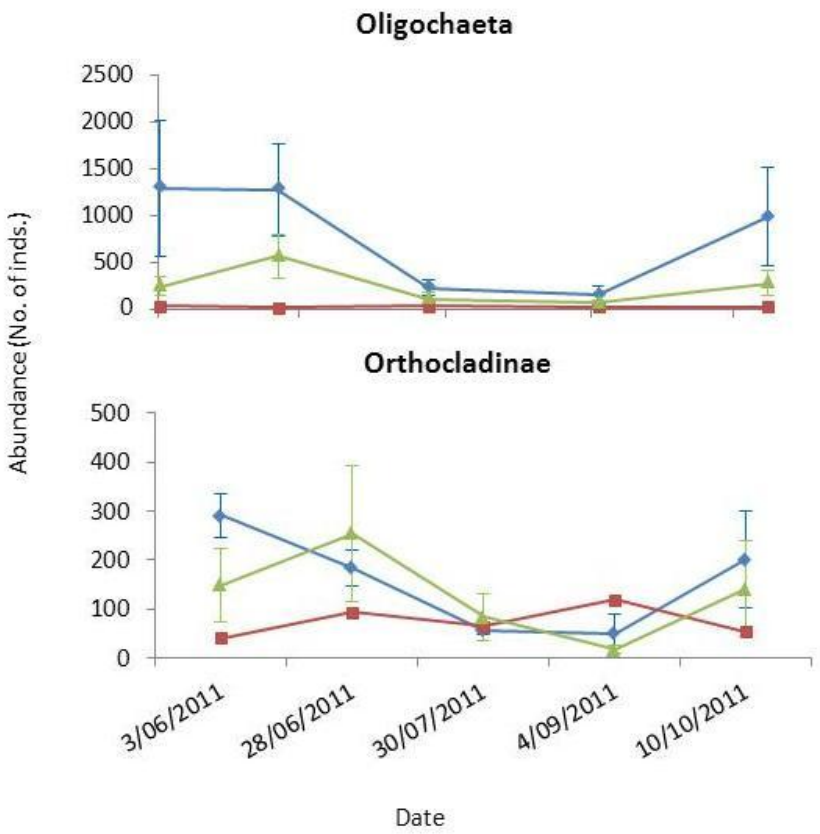



Figure 33. Mean (±1 S.E) abundance of the taxa Oligochaeta and Orthcladiinae in the Murrumbidgee (blue), Tumut (green) and Goobagandra (red) rivers over the study period.

# 4. DISCUSSION

## 4.1. Mid-Murrumbidgee Wetlands

***Water quality***

The key outcomes of the water quality monitoring component of this project are associated with understanding the impact of increased water levels on dissolved oxygen and dissolved organic carbon in the system. Importantly, while the initial organic carbon levels were high within both filled and control wetlands, the environmental release resulted in a decrease in DOC, TOC and POC levels (through dilution and exchange from the river channel) within filled wetlands without leading to any detrimental declines in dissolved oxygen levels. The potential negative impacts on the absolute dissolved oxygen concentrations due to high initial organic carbon levels were not observed due to the high solubility of dissolved oxygen and the slower rates of microbial respiration in the cold waters present at this time. By August the organic carbon levels within filled wetlands had decreased substantially from the pre-filling level and subsequent top-up flows did not lead to low dissolved oxygen events despite the warmer water temperatures. This may be an important consideration in other regions when the initial organic carbon levels are expected to be high (due to extended drying or history of low dissolved oxygen events). Therefore, watering in winter when cold temperature limit biological activity might reduce some of the short term risks of a low dissolved oxygen event and assist in protecting wetlands from low dissolved oxygen events during subsequent environmental releases or natural flood events in summer.

***Vegetation***

Aquatic and semi-aquatic vegetation cover increased significantly within wetlands that received environmental water in June 2011. In contrast there was no change in aquatic vegetation cover within the control 1 or 2 wetlands. This indicates that the environmental release was successful in promoting the recovery of aquatic and semi-aquatic vegetation within mid-Murrumbidgee wetlands. It is important to note however that aquatic vegetation cover is still relatively low compared to 2000-2004 levels and a long term watering strategy is required to allow for full recovery of aquatic vegetation communities with the mid-Murrumbidgee wetlands.

The recovery of aquatic vegetation within filled wetlands was influenced by the length of time that they had been dry prior to first filling in June 2011. Overall wetlands that received water in 2005 had a greater increase in aquatic vegetation cover than did those which had been dry for six or more years. The impacts of altered flooding histories (frequency and duration of filling events) on aquatic vegetation has been studied extensively via seed banks (Casanova and Brock 2000; Kalisz, Horth *et al.* 1997; Leck and Brock 2000; Nielsen, Podnar *et al.* ; Stromberg, Hazelton *et al.* 2009; Tuckett, Merritt *et al.* 2010). Based on seed bank studies within this region e.g. (Nielsen, Podnar *et al.*) we would normally expect relatively rapid recovery of vegetation communities from the seed bank within wetlands that had been dry for between two and 11 years. But in this study vegetation recovery was slow, with aquatic plant communities starting to only to re-establish in the second year of flooding (after approximately 15 months wet) compared to seed bank trials which typically show high levels of germination after four months (Nielsen, Podnar *et al.* 2012). However, the vegetation community within a wetland at a particular point in time typically represents a small sub set of the species that are present within the seed bank because various aspects of the hydrological regime such as timing of inundation, depth and duration influencing germination outcomes (James, Capon *et al.* 2007). While the growth and recovery of aquatic vegetation may be extremely slow within wetlands that had been subjected to extended drying, it is clear that aquatic vegetation communities are starting to recover and should recover further with subsequent years of environmental flows.

***Frogs***

Five species of frog were recorded between June 2011 and February 2012. With the exception of the Inland banjo frog which was restricted to a single wetland, the remaining four species were common and widespread, through both filled and control wetlands. The distribution of species within the mid-Murrumbidgee was similar to the Lower Murrumbidgee floodplain (Lowbidgee) which was surveyed using the same methods from 2008 to 2011 (Spencer, Wassens *et al.* 2011). However, we did not record Southern bell frogs which are relatively common in the Lowbidgee floodplain and were also recorded at Gooragool and Sunshower lagoons in November 2010 (Wassens and Amos 2011). There are a number of potential reasons why Southern bell frogs did not remain within Gooragool and Sunshower lagoon after November 2010. Firstly, the significant natural flood event which occurred in December 2010 may have contributed to the loss of this species from these two wetlands as Southern bell frogs are known to be highly dispersive during flooding (Wassens, Watts *et al.* 2008), that is the flood event gave individuals the opportunity to disperse more widely way from these wetlands. Secondly, Southern bell frogs require complex aquatic vegetation (Wassens, Hall *et al.*

2010) and the slow recovery of aquatic vegetation within these two wetlands in the first year of flooding may have reduced their suitability for Southern bell frogs, thereby preventing a population from establishing. Given the significant improvements in aquatic vegetation complexity following environmental releases in June 2011 it is possible that Southern bell frogs recolonising Gooragool and Sunshower during the March 2012 natural flood event could ultimately establish a viable population. Future monitoring of these wetlands in spring 2012 will reveal whether this has been the case.

Tadpoles were relatively widespread across the wetlands that received environmental water and within the control 2 wetlands. There was no evidence of breeding (tadpoles or egg masses) within the Control 1 probably because water levels declined throughout the study and these species typically require an increase in water levels to cue breeding. The abundance of Spotted marsh frogs and Barking marsh frogs increased over time within wetlands that received environmental water in June 2011 but not within the control 1 or control 2 wetlands which had similar abundances between years. This indicates that the environmental release contributed to an increase in the abundance of these two species. Similar increases in abundance of these two species were also recorded in the Lowbidgee floodplain following repeated environmental watering (Spencer, Thomas *et al.* 2011; Spencer and Wassens 2010). The abundance of Spotted marsh frogs and Barking marsh frogs did not increase between years within the control 2 wetlands, Turkey Flat and Yanco Ag which suggests that the populations of species within these wetlands is not increasing over time. This may be due to the hydrological regimes that occurred within these two wetlands over the study period. Both were shallow and dried out repeatedly during this study. As a result egg masses and tadpoles present in these wetlands may not have survived to metamorphosis.

***Fish***

Native species dominated fish communities within the mid-Murrumbidgee wetlands. The diversity of native fish was higher than previously recorded by (Gilligan 2005) within the mid-Murrumbidgee wetlands but was lower than Lowbidgee floodplain where nine species were recorded following environmental flooding between 2008-2010 (Spencer, Thomas *et al.* 2011; Spencer and Wassens 2009). Juveniles were recorded for all native species, with the proportion of juveniles increasing over time; this indicates that the breeding occurred within the wetlands rather than as a result of individuals moving into the wetlands.

Carp and goldfish are very widespread and abundant throughout the Murray-Darling Basin and these two species were widespread through wetlands in the mid-Murrumbidgee. However the breeding response of these species was lower in the mid-Murrumbidgee following the environmental release than has been observed across the Lowbidgee floodplain (Spencer and Wassens 2009).

Oriental weatherloach was recorded in the mid-Murrumbidgee wetlands for the first time; this species was also recorded in the Lowbidgee floodplain in 2011 and appears to have expanded its range following the natural flood event in December 2010. Only adult weatherloach were recorded with no juveniles recorded in surveys after August 2011 which indicates that this species was not breeding successfully within the filled wetlands. Studies of this species in the USA have shown that it can survive for up to 81 days in a desiccated state with very low soil moisture (Koetsier and Urquhart 2012). If this species becomes established within individual wetlands, longer drying periods > 80 days between watering events could be used to reduce numbers, although the negative impacts of drying on native fish and freshwater turtles would also need to be considered.

***Waterbirds***

Waterbird communities in the mid-Murrumbidgee wetlands were diverse with 36 species recorded during the 2011-12 surveys. Waterbird responses to the flooding were linked to seasonal effects, with greater numbers of waterbirds observed in all the surveyed wetlands during spring. Top-up flows in summer 2011 provided greater habitat for waterbirds and extended the availability of habitat and some breeding into February 2012.

Overall, colonial waterbird breeding was limited with only small numbers of nests recorded in Gooragool, Yarrada and McKennas lagoons. This is not unexpected given this is only the second year of filling of the mid-Murrumbidgee wetlands and there being significant and widespread natural flooding across many regions within of the Murray-Darling Basin in 2010-12.

Surveys to date have indicated that the mid-Murrumbidgee wetlands are in a recovery stage with re-establishing wetland vegetation communities following the 2011 environmental release. Most waterbirds prefer wetland habitats with aquatic vegetation, either as foraging habitat or to provide shelter from weather and predators (Scott, 1997). It is likely that the wetlands will become more attractive to waterbirds as wetland vegetation becomes further established. Further surveys over successive years of flooding are needed to allow for a comprehensive assessment of waterbird use of the mid-Murrumbidgee wetlands.

***Turtles***

River regulation and extended drought across the Murray-Darling Basin has reduced the temporal and spatial availability of wetland habitats for freshwater turtles. Recent surveys have shown that turtle populations are severely depleted in the Murrumbidgee (Spencer and Wassens, 2010) and Murray catchments (Chessman 2011). Small numbers of freshwater turtles were caught in the mid-Murrumbidgee wetlands during 2011-12, and it was an encouraging sign to see small numbers of recently hatched Macquarie River turtles.

Macquarie River turtles were detected in Yarrada lagoon which is located in close proximity to the river channel, and Molleys lagoon, a deep relatively permanent channel. This distribution matches habitat use observed in turtle populations in the Murray (Chessman 1988) and Lower Murrumbidgee wetlands (Spencer and Wassens, unpublished data). Compared to long-necked turtles, Macquarie River turtles are much more vulnerable to the effects of drying of temporary wetlands, because long-necked turtles are able to disperse overland when wetlands dry down (Chessman 2011). However, both species are likely to benefit greatly during periods of connection between the river and the floodplain, with growth rates in turtles often higher in wetlands that have a drying phase (White 2002).

As recruitment has been limited in the mid-Murrumbidgee wetlands over the last decade it will take considerable time for freshwater turtle populations to re-establish. As with frog, native fish and waterbird populations, successive years of flooding the mid-Murrumbidgee wetlands will be required to restore turtle populations in this region. Flooding wetlands closest to the Murrumbidgee River and other permanent waterbodies would be highest priority during dry periods to promote turtle dispersal and provide adequate food resources. However, more detailed surveys are needed for a comprehensive assessment of turtle distribution in the mid-Murrumbidgee wetlands.

## 4.2. In–stream environment

The environmental flow had a positive effect on the in-stream ecosystem by significantly reducing the biomass of biofilm at Murrumbidgee sites 1 and 2, most likely due to scouring of biofilms from increased water velocity. In the Murrumbidgee River, the environmental flow provided benefits by increasing the relative dominated by early succesional algae (e.g. diatoms). In the Tumut River the environmental flow scoured the green algae present at Tumut sites 1, 2 and 3. Scouring of biofilm provides benefit by contributing nutrients and food into the water column, thus providing an important resource for downstream communities. It also provides benefit to the community by reducing the nuisance factor that occurs when the biofilms form mats and builds up to unacceptable levels.

These findings are consistent with studies of biofilm responses to variable flow releases from Dartmouth Dam. It has been demonstrated that pulsed flows can scour and reset biofilms; reducing biomass, reducing the dominant taxa and enabling early successional diatoms to become established, facilitating a shifting in the biofilm community towards that of the reference stream (Watts, Nye *et al.* 2005; Watts, Ryder *et al.* 2008; Watts, Zander *et al.* 2011).

In the upstream reaches of the Tumut River, the environmental flow provided only short-term benefits because dam operations returned to normal regulated practices soon after the environmental flow. Approximately five weeks after the recession of the environmental flow the proportion of green algae had increased significantly at Tumut sites 1 and 2 and it increased further again by 4/9/2011. Similarly, five weeks after the environmental flow the biomass of biofilm at some of the most upstream sites had significantly increased to levels higher than those observed prior to the environmental flow. The biofilm biomass was very high at several sites (e.g. Tumut site 1) was higher than any previously recorded levels downstream of Dartmouth Dam in the Mitta Mitta River (Watts, Nye *et al.* 2005; Watts, Ryder *et al.* 2008; Watts, Zander *et al.* 2011). The biomass of biofilm at many sites greatly exceeded the recommendations of (Quinn 1991) which provides guidelines for the control of undesirable biological growths in water, and states that biomass should not exceed 100mg chlorophyll-a /$m^2$. The chlorophyll-a levels observed after the environmental flow in the Murrumbidgee and Tumut Rivers were extremely high when compared to levels previously observed downstream of Dartmouth Dam in the regulated Mitta Mitta River (Watts, Nye *et al.* 2005; Watts, Ryder *et al.* 2008; Watts, Zander *et al.* 2011)

The environmental flow provided a longer-term benefit at Tumut sites 3, and 4, downstream of the confluence with the unregulated Goobarragandra River, because the unregulated flows from the Goobarragandra River contributes to the health of these downstream reaches. The biofilm biomass and Chlorophyll-a levels at the two downstream sites in the Tumut River (sites 4 and 5) did not increase to the same extent as the upstream sites. The environmental flow had a short-term benefit on the in-stream ecosystem by increasing in the number of macroinvertebrate taxa in the Tumut River immediately after the environmental flow. However, the total number of taxa reduced after dam operations returned to normal regulated practices. The responses of macroinvertebrate taxa were predictable based on previous studies. Taxa which increased in abundance immediately after the environmental were those that rely on fast flows for feeding (e.g. Hydropyschids and Simulids) and have the capacity to recolonise sites through downstream drift. Other taxa which are associated with low flows (e.g. Corixids) showed a decrease in abundance after the flow, suggesting suitable low flow habitats were less available during the time of the flow pulse, but once such habitats were established after the pulse, numbers increased. This short-term response of macroinvertebrates has also been observed in the Mitta Mitta River downstream of Dartmouth Dam following a variable flow release (Sutherland, Ryder *et al.* 2002). For majority of the dominant macroinvertebrate taxa there was a short term increase in abundances immediately after the flow pulse, followed by a longer term trend of abundances reducing back to a level somewhat similar to that observed prior to the environmental flow.

These findings suggests that while the environmental flow provided a short-term benefit to the in-stream ecosystem through scouring of biofilms, the benefits were not sustained when the dam operations returned to normal regulated practices. These findings are consistent with observations in the Mitta Mitta River. One of the new interim guidelines developed for the operation of Dartmouth Dam to improve the in-stream ecosystem (Watts, Ryder *et al.* 2009) is to "Include pulses of different magnitudes (over a period of months) that reflect the variability of a more natural flow hydrograph" so that the benefits of pulsed flows can be sustained over time. In the Murrumbidgee River the environmental flow provided a larger flow pulse that would be delivered at this time of year during normal river operations.

The benefits of the environmental flow were short lived in the most upstream sites in the most upstream sites in the Tumut River. This is not surprising because since 2005 the flow regime in the Tumut River has been more highly regulated that the Murrumbidgee River. Furthermore, immediately after the environmental flow there were several regulated releases from Blowering Dam fluctuating from baseflows up to approximately 8000 ML/day (almost bankful discharge), with one event in July continuing at this discharge for approximately three weeks. This demonstrates the need for environmental flows of different magnitudes in these regulated systems because normal dam operations do not provide the range of flow variability.

# 5. SYNTHESIS

This monitoring program assessed ecological responses within wetland and in-stream habitats during the June 2011 environmental release into the Murrumbidgee. The wetlands of the mid-Murrumbidgee have been subject to an extended period of drying between 2000 and 2009, which far exceeded their natural wetting and drying cycles. Therefore, the impact of the environmental release needs to be considered in the context of wetland recovery. The environmental release led to a number of positive changes within the mid-Murrumbidgee including a reduction in organic carbon levels, which was achieved without leading to low dissolved oxygen levels in the wetlands or river. The reduced carbon levels increased the cover of aquatic vegetation that, in time, should assist in rebuilding wetland seed banks and promote breeding by native frogs, fishes, waterbirds and turtles.

The benefits of the environmental release were extended due to subsequent natural and managed releases in spring and summer 2011. These subsequent flows increased the connectivity between the wetlands and the river and also led to increases in water levels that are likely to have promoted repeat breeding by frogs, some native fish species, such as bony bream, and waterbirds. On their own, these small flows in spring and summer may not have had a significant impact on wetland flora and fauna, but, when combined with the large June 2011 environmental release, the overall outcomes were positive. For example, it was clear that the wetlands receiving water during the December 2011 release maintained an aquatic habitat through summer that enabled a higher diversity and abundance of waterbirds. As the mid-Murrumbidgee wetlands are still in a recovery phase, it is extremely important that management intervention, including environmental watering, is maintained. Recovery of these wetlands should be a key priority for future watering. Without a long-term commitment to the health and recovery of these nationally important wetlands, the gains created by the June 2011 release could be lost in future years.

Within the river channel (in-stream) the environmental flow in the Murrumbidgee and Tumut rivers had a shorter term benefit when compared to the wetlands. That is, while it provided a short to medium term benefit through scouring of biofilms, this benefit could not be sustained within the Tumut River once dam operations returned to normal regulated practices. Despite this, these results suggest it is possible to deliver environmental flows from Burrinjuck and Blowering Dams that benefit in-stream ecosystems in the reaches immediately downstream of the dams.

In-stream benefits could be improved by the delivery of environmental flows that explicitly consider in-stream outcomes during water delivery. By using a careful release strategy, future environmental releases may be used to optimise benefits for both in-stream and wetland ecosystems, using a similar volume of water.

# 6. RECOMMENDATIONS

The mid-Murrumbidgee wetlands are currently in a recovery phase. Aquatic vegetation diversity and cover while increasing, have yet to recover to pre-drought levels. Populations of native fish, frogs, waterbirds and turtles are still comparably small compared to wetlands in the Lower-Murrumbidgee. In this context the next five years of watering will be critical for the successful recovery of wetland communities. The key recommendations are as follows:

**Thresholds for vegetation recovery**

- Vegetation monitoring showed a clear relationship between the recovery of aquatic vegetation and the length of time that the wetland had been dry. The wetlands of the mid-Murrumbidgee historically filled annually on all but the driest years. Wetlands that had been dry for more than five years had very poor rates of recovery compared to those which had been dry for between three to five years. It is critical that future water management does not allow wetlands to remain dry for periods that far exceed their normal flooding regime. In this instance wetlands tolerated being dry for two to three years but dry periods greater than this had a negative impact on vegetation communities.

**Timing of flows**

- Watering wetlands in winter can reduce dissolved organic carbon levels without leading to a low dissolved oxygen event. The lessons learnt from positive impact of winter watering could be extended to other regions in the southern Murray-Darling Basin where there is clear evidence that risk factors predicting low dissolved oxygen events are present, including high organic carbon loads and a history of low-dissolved oxygen events.

- Other benefits from watering wetlands in winter and early spring is that it provides cues to waterbird species that typically commence breeding in spring and summer months, but require a lag period to build up their fat reserves before laying of eggs (Scott, 1997). Ideally nest trees should be watered for five to ten months for successful breeding, and then be allowed to dry back before the next watering. This drying period is needed to maintain the health of the river red gums, which should not normally be flooded continuously for more than 18 months, as well as maintaining the productivity of the wetland as foraging habitat for waterbirds (Briggs and Thornton 1999).

**Top-up flows**

- Spring and summer top-up flows promoted the recovery of native fish communities within wetlands by allowing for movement in and out of wetlands and providing cues for breeding, they were also important in promoting frog breeding, maintaining waterbird habitat and maintaining aquatic vegetation.

  Small top-up flows may be particularly important in situations when the bulk of water is delivered in winter and should be explicitly integrated into future watering strategies within the Murrumbidgee Catchment.

**Optimising in-stream and wetland benefits**

- The environmental release had a positive impact on both wetland and in-stream ecosystems. However the in-stream benefits of the environmental release were far greater in the Murrumbidgee River than in the Tumut River. This occurred because the environmental release from Blowering Dam to the Tumut River had a similar river height and discharge to normal delivery for consumptive use. This highlights the importance of understanding the normal operational flow regimes that are present within a regulated river system and understanding of the key flow components that reflect the variability of a more natural flow hydrograph. Environmental releases should specifically aim to restore key flow components that have been lost through river regulation and avoid delivering flows that would exacerbate negative impacts of regulated operations. By using a careful release strategy, environmental flows can be delivered to optimise benefits for both wetland and in-stream ecosystems.

- In the absence of detailed flow analysis for the Murrumbidgee and Tumut Rivers it is advisable when delivering environmental water from Burrinjuck and Blowering Dams to consider one of the new interim guidelines developed for the operation of Dartmouth Dam to improve the in-stream ecosystem (Watts, Ryder *et al.* 2009) which is to "*Include pulses of different magnitudes (over a period of months) that reflect the variability of a more natural flow hydrograph*".

**Monitoring**

Continued monitoring of the Murrumbidgee system is critical as it provides timely advice to short term management interventions (e.g. follow-up watering) while contributing to the refinement and improvement of the long-term environmental flows management strategy. Within the mid-Murrumbidgee wetlands we monitored fish, frogs, waterbirds and turtles populations as well as vegetation, water quality and carbon over an eight month period. This information will allow Commonwealth and NSW water managers to make informed decisions for future watering actions (see below).

Assessing the response of **key components of wetland ecosystems** is important because individual groups of taxa often respond to different aspects of the flood pulse and over differing spatial and temporal scales (e.g., long and short term flooding regimes). Flooding regimes that benefit one group may have a negative impact on other groups. For example, an increase in wetland permanence may have a positive impact on some native fish species but can have negative impact on frog populations and aquatic vegetation.

- **Aquatic and semi-aquatic vegetation communities** can be easily assessed in the field by an experienced practitioner and were a very useful indicator of past flooding regime and wetland recovery. Compared to vertebrates, like fish and frogs, the response of aquatic vegetation following an environmental flow is closely linked to local condition within individual wetlands, which is why they are more sensitive to past flooding regimes than other indicators. In this respect aquatic vegetation monitoring provides valuable insights into the longer-term recovery of wetlands within the mid-Murrumbidgee.

- **Fish and tadpoles** were particularly useful indicators in this study because both exhibited a clear response, i.e., change in abundance, size structure (fish) and development stage (tadpoles) to each individual environment release. Unlike aquatic vegetation the response of fish is strongly linked to connectivity between the river and wetland particularly the number and timing of top-up flows. Like vegetation, fish and frog populations responses can be assessed in the field and the outcomes of monitoring can be rapidly integrated into adaptive management within the same water year. For example, in 2011-12 the outcomes of monitoring highlighted the need for additional top-up flows ( see section below)

- Assessment of **frog populations** was valuable for larger-scale monitoring of wetlands, including the control 2 wetlands, because it could still be conducted when wetlands were too shallow to allow for intensive fish and tadpole surveys.

- **Waterbirds** are good indicators of large-scale wetland availability and the impact of flooding duration. It was particularly important to maintain water levels in the mid-Murrumbidgee wetlands and monitor breeding to ensure nesting cycles were completed over the summer months. The mid-Murrumbidgee region formerly provided major breeding habitat for nesting colonial waterbird species, including darters, cormorants, herons, egrets, white ibis and spoonbills (Briggs and Thornton 1999). Ongoing monitoring will be required to document the recovery of waterbird populations in this region, and when breeding does occur, to provide advice on follow-up watering needed in spring and summer to maintain waterbird breeding in key locations.

- **Dissolved organic carbon** can be an important risk factor for low dissolved oxygen events and as such it is useful to monitor dissolved organic carbon prior to, during and immediately after an environmental release. This enables allows managers to assess the risk of low dissolved oxygen events and plan for management interventions such as dilution flows.

- **Biofilms** were a useful indicator of the **in-stream response** to the environmental flow because they had predictable responses to flow and respond over a short time frame (days to weeks) which is appropriate for assessing the immediate responses to a flow event, and they reflect broader changes in ecosystem health (e.g. food resources). In systems where biofilms may have grown to levels that are unacceptable to the community, monitoring the changes to biofilm biomass can also provide an indication of benefits to recreational amenity of the river following environmental flows.

- **Freshwater macroinvertebrates** are generally good indicators of river health and flow management because their taxonomy is well known and many taxa are sensitive to stress and respond to changes in environmental conditions. However, assessment of macroinvertebrate communities can be very labour intensive and, from a cost-benefit perspective, biofilms provide a more predictable and rapid assessment of the short-term in-stream benefits of an environmental flow event.

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# APPENDICES

# Appendix 1: Oxygen saturation data for individual river and wetland sites

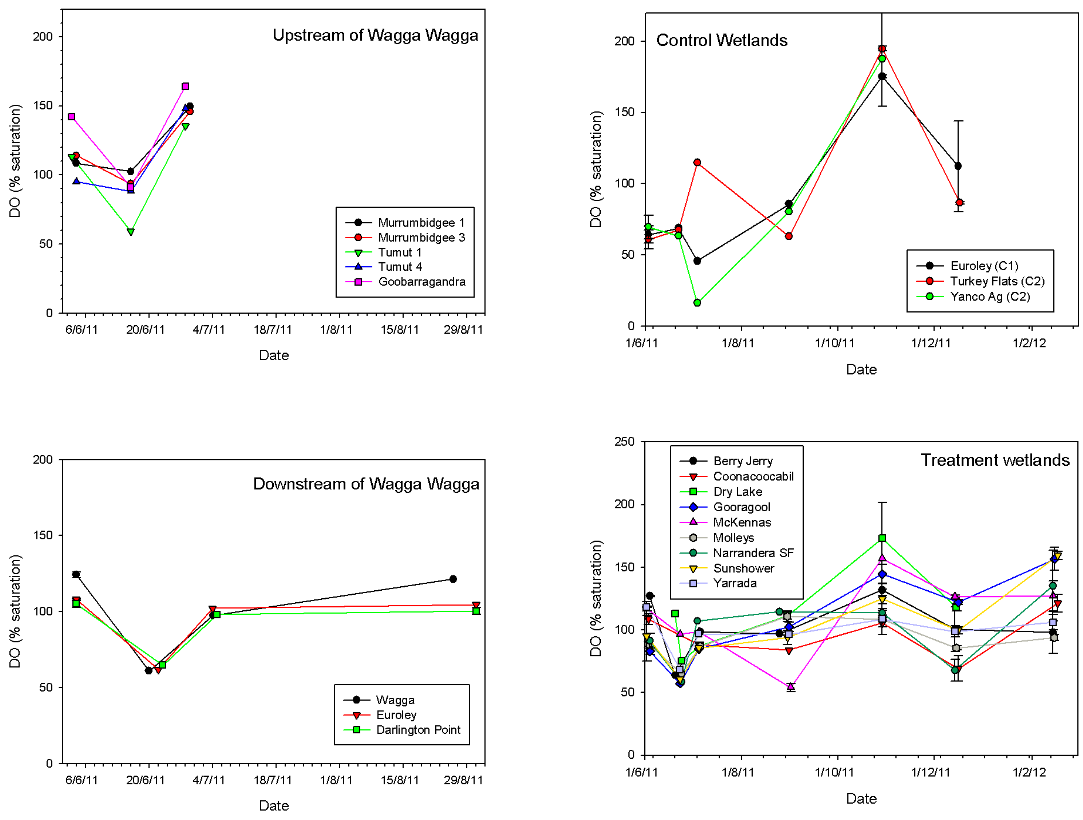


Figure A1.1 Trends in percentage oxygen saturation over time at river sites and wetlands.

## Appendix 2: Spectroscopy

With the additional natural watering of the wetlands in late August/September, it was decided that some ongoing monitoring of the wetland organic carbon would be undertaken beyond the original planned sampling period. Analysis of the organic carbon characteristics using absorbance and fluorescence spectroscopy had been undertaken on all sampling dates and this was continued through to February. Absorbance spectroscopy, in addition to providing some information on the nature of the organic carbon present, can also be used to approximate the dissolved organic carbon concentration (where the mixture of compounds remains similar). Figure A2.1 illustrates the relationship between the absorbance of ultraviolet light (250 nm) by 1cm of the water and the dissolved organic carbon concentration for all samples where DOC was measured for this project. The scatter in the data indicates some variation in the organic carbon mixture, but there is a clear correlation between the two measurements. Absorbance at 250 nm can be used to give an indication of the organic carbon concentration for samples collected in October, December and February, however, it should be noted that for absorbances well above 1 caution should be exercised as non-linear behaviour is more likely in this region.

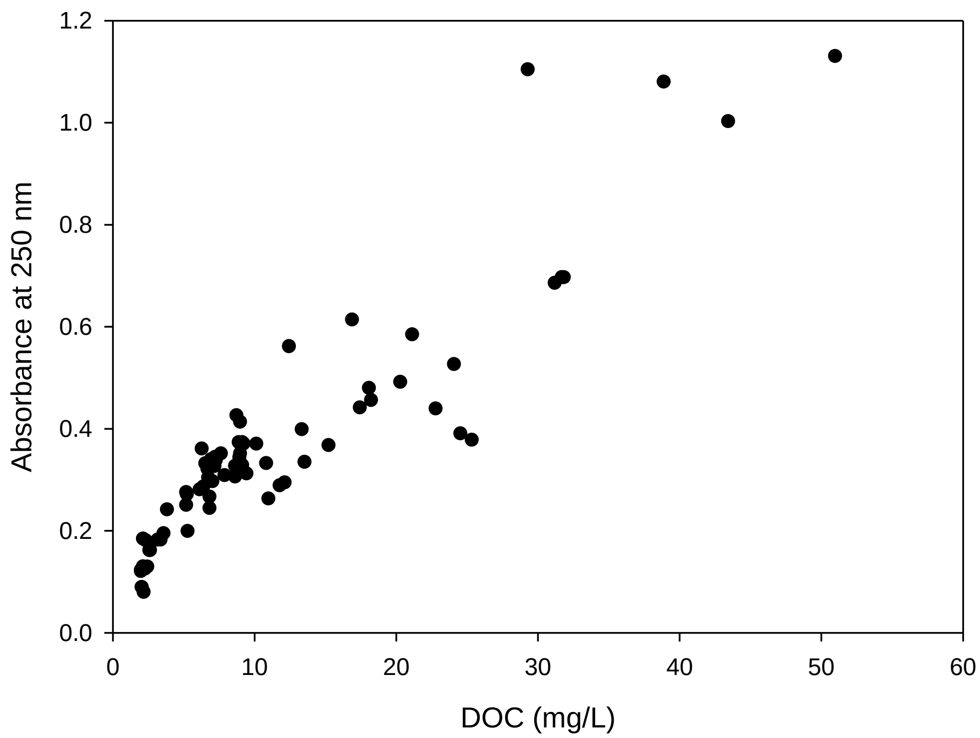


Figure A2.1 Relationship between measured absorbance at 250nm and dissolved organic carbon concentration for samples collected June-August 2011.

The shape of an absorbance scan can give an indication as to the distribution of sizes of molecules making up the mixture of dissolved organic carbon. The ratio of absorbance at 250 nm to 340 nm can be used to indicate changes in average molecular size, with larger molecules increasing the absorbance at longer wavelengths i.e. broader absorbance scans indicate larger size molecules than steep increases where both scans have similar absorbances at 250nm (Howitt et al, 2008). Absorbance scans for each of the sampling dates are given below. Note the general similarity of the treatment wetlands prior to the environmental water release in June with the exception of McKennas. Euroley is also similar to the treatment wetlands but the C2 wetlands are initially quite different to the rest of the sites. During the flow pulse in June the treatment wetlands generally reflect the properties of the organic matter in the river, again with the exception of McKennas. In August the organic matter in the inflow to Gooragool wetland from the channel exhibits clear differences to all the other sites, indicating a different origin of that organic matter.

The Euroley control wetland remains quite similar to the treatment wetlands through to August, but in October and December it can be seen to separate from the other wetlands as it moves through a concentration/drying phase. In December the wetlands which received a top-up flow are grouping together and separating from the other wetlands (with the exception of Coonacoocabil). By February Sunshower separated markedly from the rest of that grouping, suggesting either a different rate of organic matter input or a different rate of drying.

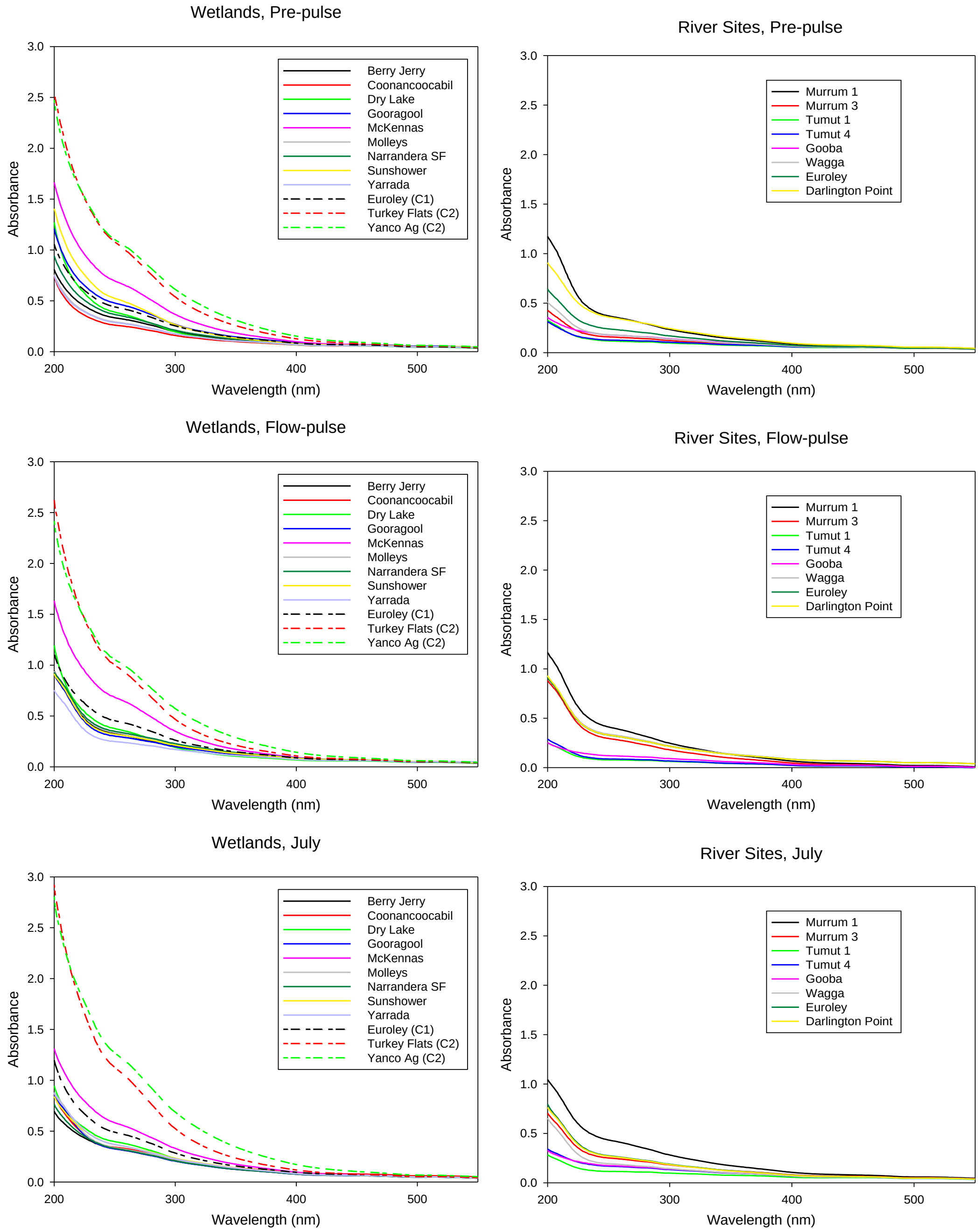


Figure A2.2. Absorbance scans of dissolved organic matter from wetland and river sites for each sampling date (cont. on following page).

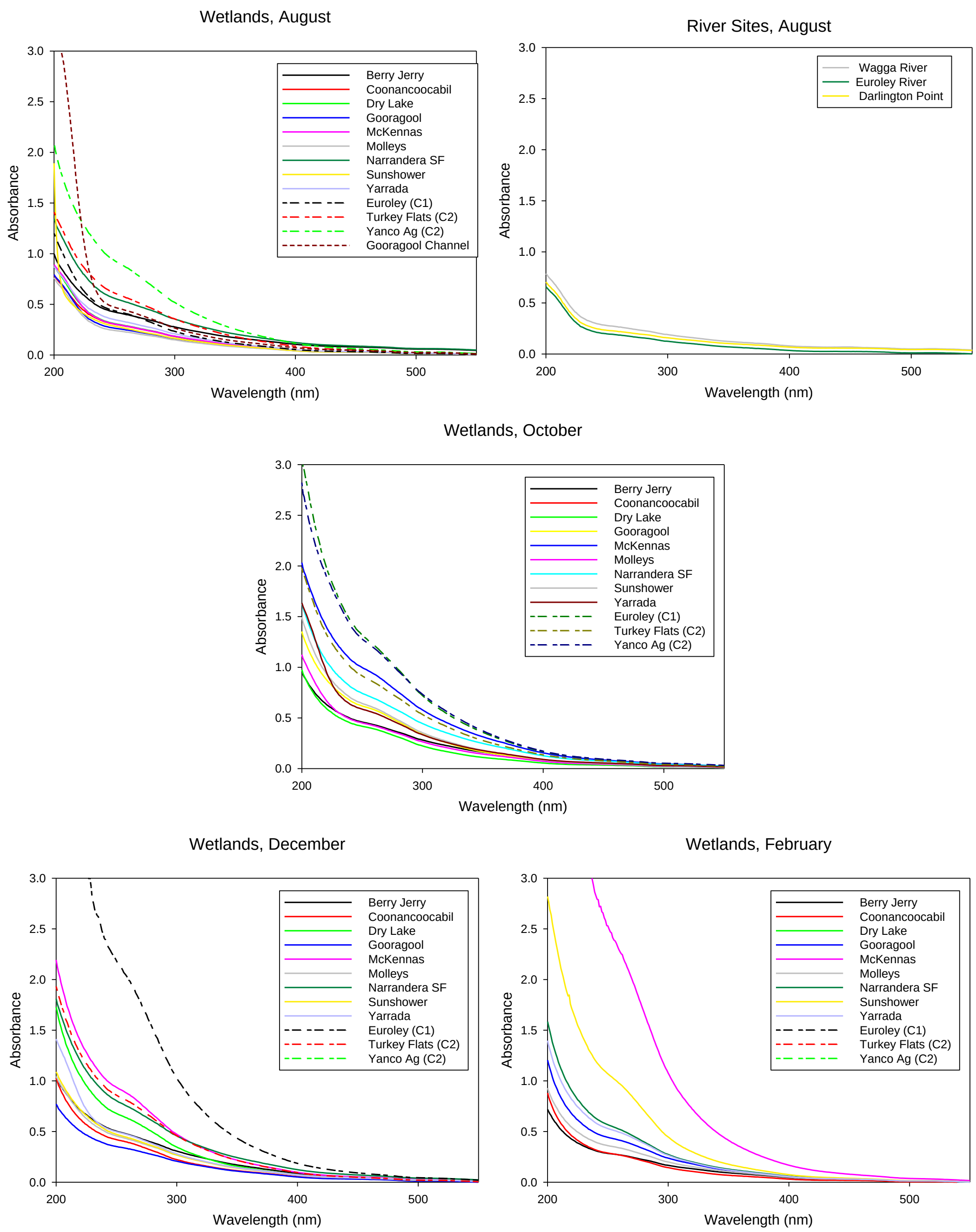


Figure A2.2. Absorbance scans of dissolved organic matter from wetland and river sites for each sampling date (cont.).

## Excitation Emission Spectroscopy

Further characterization of the organic matter was undertaken using Excitation Emission Spectroscopy (a variety of fluorescence spectroscopy). The 3-D fluorescence spectra (or Excitation-

Emission Matrices) of dissolved organic matter provides information on the average characteristics of the component molecules. A very large number of compounds are present in natural waters and all fluorescing materials (generally large, complex structures) will contribute to the measured spectrum. Changes to the peak positions can give an indication to changes to major components of the dissolved organic matter (Coble 1996).

This technique can be thought of as a fingerprinting technique, which gives a broad indication of the relative proportions of aromatic proteins, humic and fulvic substances. This technique is useful for tracking microbial 'aging' and photochemical degradation of the organic matter (Howitt et al 2008).

The plots have the excitation wavelength on the y-axis. This is the wavelength of light shone into the sample by the instrument. On the x-axis is the emission wavelength; that is, the wavelength of light given off by the sample at right angles to the incoming beam. The intensity of the fluorescence (how much light is given off, corrected for absorbance by the sample) is represented by the colours of the contour plot, with more intense fluorescence represented by the blue end of the scale.

Broadly, as a guide to the interpretation of the contour plots, an increase in fluorescence towards the right of the plot and towards the top of the plot indicates an increasing humic content, i.e. larger, more complex molecules and generally lower carbon bioavailability (Howitt et al, 2008). The two diagonal lines are artefacts of the technique and will be present in all samples. Where available, the measured DOC concentration for the sample is included with the fluorescence plots. It is evident that changes in DOC concentration do not always result in a corresponding change in humic signature, especially at some wetland sites. This data is evidence of a change in the mixture of organic carbon over time and associated with the connection between the river and the wetlands. Processes that may result in a change in the organic carbon composition include: the leaching of additional dissolved material from floodplain vegetation, litter and soils; the microbial processing of dissolved organic matter; and photochemical changes induced by exposure to sunlight (Howitt et al, 2008).

Prior to the environmental water release the river sites show a clear trend towards increasing humic character of the organic matter as you travel downstream, with the exception of the Murrumbidgee 1 site, which has a highly humic character throughout June. Sites immediately downstream, however, are more heavily influenced by the Tumut and Goobarragandra discharges, which have very little fluorescence signal. The Murrumbidgee 1 site has a much larger influence on the downstream sites during the environmental water release in June. During July and August the fluorescence signal returns to the previous pattern, with more local influence during the natural flow event in August.

In the wetland fluorescence results there are a number of key points to highlight. Note that on some occasions the absorbance of the sample was sufficiently high in the ultraviolet region to result in artefacts in the fluorescence results, despite the application of absorbance corrections. Where this was anticipated to be a problem, diluted samples were analysed. These samples are used to indicate the location of organic matter peaks, but it should be noted that the intensity of those peaks has been reduced by the dilution process. The treatment wetlands and Euroly have in common the presence of a small peak at shorter wavelengths which gradually disappears from June to August. The changes in carbon composition over time vary with wetlands. At Berry Jerry the June filling results in some loss of humic character (through either dilution or exchange with the river) and an increase in fluorescence at shorter wavelengths and the organic matter profile at this time reflects that of the river water at Wagga Wagga, upstream. The two later top-ups in August and December result in an increase in humic signature in this wetland. This pattern can also be seen in some of the other wetlands. For most of the wetlands there is a decrease in fluorescence between December and February, however, the opposite is true for the Sunshower wetland (consistent with the change in absorbance scans over this period). It is noted that both of the C2 wetlands started with very different organic matter profiles to the other wetlands and this is maintained throughout the sampling period and is consistent with differences in other water quality parameters e.g. ammonia.

# Appendix 3: Nutrients

Table 6. Ammonia concentrations (mg N/L).

| | Site | Pre release | Peak | Post release | August |
|---|---|---|---|---|---|
| **Filled wetlands** | Berry Jerry | <0.2 | <0.2 | <0.2 | <0.2 |
| | Coonancoocabil | <0.2 | | <0.2 | <0.2 |
| | Dry Lake | <0.2 | <0.2 | <0.2 | <0.2 |
| | Gooragool | <0.2 | <0.2 | <0.2 | <0.2 |
| | McKennas | <0.2 | <0.2 | <0.2 | 0.2 |
| | Molleys | <0.2 | <0.2 | <0.2 | <0.2 |
| | Narrandera SF | <0.2 | <0.2 | <0.2 | <0.2 |
| | Sunshower | <0.2 | <0.2 | <0.2 | <0.2 |
| | Yarrada | <0.2 | <0.2 | <0.2 | <0.2 |
| **Control** | Euroley (C1) | <0.2 | <0.2 | <0.2 | <0.2 |
| | Turkey Flats (C2) | 4.6 | 3.3 | 2.9 | <0.2 |
| | Yanco Ag (C2) | 0.9 | 1.1 | 1.7 | <0.2 |
| **Murrumbidgee River at** | Wagga | <0.2 | | <0.2 | <0.2 |
| | Euroley | <0.2 | <0.2 | <0.2 | <0.2 |
| | Darlington Point | <0.2 | <0.2 | <0.2 | 0.3 |
| | Blanks | <0.2 | | | |

Table 7. Nitrate and Nitrite (NOx) concentrations (mg N/L).

| | Site | Pre release | Peak | Post release | August |
|---|---|---|---|---|---|
| **Filled wetlands** | Berry Jerry | <0.2 | 0.4 | <0.2 | <0.2 |
| | Coonancoocabil | <0.2 | | <0.2 | <0.2 |
| | Dry Lake | 0.4 | 0.6 | <0.2 | <0.2 |
| | Gooragool | 0.2 | 0.5 | 0.2 | <0.2 |
| | McKennas | <0.2 | <0.2 | <0.2 | <0.2 |
| | Molleys | <0.2 | 0.4 | 0.3 | <0.2 |
| | Narrandera SF | <0.2 | 0.4 | <0.2 | <0.2 |
| | Sunshower | <0.2 | 0.4 | <0.2 | <0.2 |
| | Yarrada | <0.2 | 0.5 | 0.2 | <0.2 |
| **Control** | Euroley (C1) | <0.2 | <0.2 | <0.2 | <0.2 |
| | Turkey Flats (C2) | <0.2 | <0.2 | <0.2 | <0.2 |
| | Yanco Ag (C2) | <0.2 | <0.2 | <0.2 | <0.2 |
| **Murrumbidgee River at** | Wagga | 0.2 | | <0.2 | <0.2 |
| | Euroley | <0.2 | 0.4 | 0.4 | <0.2 |
| | Darlington Point | 0.3 | 0.4 | 0.3 | 0.6 |
| | Blanks | <0.2 | | | |

Table 8. Total Kjeldahl Nitrogen (mg N/L).

| | Site | Pre release | Peak | Post release | August |
|---|---|---|---|---|---|
| **Filled wetlands** | Berry Jerry | <2 | <2 | <2 | <2 |
| | Coonancoocabil | <2 | | <2 | <2 |
| | Dry Lake | <2-3 | 3 | <2 | <2 |
| | Gooragool | <2-5 | <2 | <2 | <2 |
| | McKennas | <2 | 3 | <2 | <2 |
| | Molleys | <2 | <2 | <2 | <2 |
| | Narrandera SF | <2 | <2 | <2 | <2 |
| | Sunshower | <2 | <2 | <2 | <2 |
| | Yarrada | <2 | <2 | <2 | <2 |
| **Control** | Euroley (C1) | <2 | 3 | <2 | 5 |
| | Turkey Flats (C2) | 7 | 8 | 4 | <2 |
| | Yanco Ag (C2) | 3 | 5 | 3 | 3 |
| **Murrumbidgee River at** | Wagga | <2 | | <2 | <2 |
| | Euroley | <2 | <2 | <2 | <2 |
| | Darlington Point | <2 | <2 | <2 | <2 |
| | Blanks | <2 | | | |

Table 9. Dissolved phosphate (mg P/L).

| | Site | Pre release | Peak | Post release | August |
|---|---|---|---|---|---|
| **Filled wetlands** | Berry Jerry | 0.01 | 0.02 | 0.04 | 0.01 |
| | Coonancoocabil | <0.01 | | 0.04 | 0.01 |
| | Dry Lake | 0.02 | 0.01 | 0.01 | 0.04 |
| | Gooragool | 0.02 | 0.02 | 0.07 | 0.04 |
| | McKennas | 0.01 | 0.01 | 0.04 | 0.01 |
| | Molleys | 0.01 | 0.03 | 0.07 | 0.01 |
| | Narrandera SF | 0.01 | 0.03 | 0.03 | 0.01 |
| | Sunshower | <0.01 | 0.02 | 0.04 | 0.01 |
| | Yarrada | 0.01 | 0.02 | 0.04 | 0.01 |
| **Control** | Euroley (C1) | 0.02 | 0.04 | 0.08 | <0.01 |
| | Turkey Flats (C2) | 0.03 | 0.03 | 0.08 | 0.01 |
| | Yanco Ag (C2) | 0.02 | 0.02 | 0.09 | 0.01 |
| **Murrumbidgee River at** | Wagga | 0.01 | | 0.02 | 0.01 |
| | Euroley | 0.02 | 0.02 | 0.02 | <0.01 |
| | Darlington Point | 0.03 | 0.02 | 0.04 | <0.01 |
| | Blanks | <0.01-0.01 | | | |

# Appendix 4. Waterbird species in the mid-Murrumbidgee wetlands, Jun 2011-Feb 2012

Table 10. Mean abundance of waterbird species in the mid-Murrumbidgee wetlands, Jun 2011-Feb 2012.

| Common name | Berry Jerry | Coonacoocabil | Dry Lake | Gooragool | McKennas | Molleys | Narrandera SF | Sunshower | Yarrada | Euroley (C1) | Turkey Flats (C2) | Yanco Ag (C2) | % Occurrence* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Australasian grebe | 2.3 | 2.7 | 0 | 0.8 | 1.8 | 2.0^ | 1.3 | 1.0 | 1.7 | 0.5 | 3.0 | 1.3 | 92 |
| Australasian shoveler | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7.0 | 0 | 8 |
| Australian pelican | 0 | 0 | 28.3 | 0 | 48.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 17 |
| Australian shelduck | 0 | 0 | 0 | 2.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 |
| Australian white ibis | 0 | 3.5 | 0.8 | 3.6 | 1.2 | 0.6 | 0.8 | 0 | 0.5 | 0.2 | 0.2 | 0.2 | 83 |
| Australian wood duck | 0.8 | 50.4 | 4.0 | 57.0 | 23.8 | 7.7 | 8.0 | 24.0 | 26.6 | 1.7 | 24.7 | 4.3 | 100 |
| Black swan | 0 | 0 | 0.5 | 0.8 | 0 | 0 | 0 | 0.5 | 0 | 0 | 0.5 | 0 | 33 |
| Black-fronted dotterel | 0 | 0 | 1.0 | 0 | 0 | 0 | 0 | 3.0 | 2.7 | 0 | 0 | 0 | 25 |
| Black-tailed native hen | 0 | 0 | 3.0 | 1.0 | 2.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 25 |
| Chestnut teal | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 7.0 | 0 | 0 | 0 | 0 | 8 |
| Darter | 0.8 | 0.8 | 3.4 | 6.8^ | 3.0^ | 0.5 | 0.2 | 2.2 | 14.0 | 0.2 | 0.3 | 0 | 92 |
| Dusky moorhen | 1.3^ | 0 | 0 | 0.3 | 0 | 0 | 0.7 | 0.3 | 0 | 0 | 0 | 2.8 | 42 |
| Eurasian coot | 0.4 | 0 | 0 | 4.6 | 0.2 | 0 | 0.2 | 0.2 | 0 | 0 | 0 | 0.8 | 50 |
| Glossy ibis (C) | 0 | 0 | 0 | 0 | 32.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 |
| Great cormorant | 0.3 | 0.8 | 4.0 | 11.0^ | 16.4 | 0 | 0 | 6.8 | 9.8^ | 0 | 0 | 0 | 58 |
| Great egret (J, C) | 0.2 | 0 | 2.0 | 1.5 | 1.3 | 2.8 | 0.4 | 3.0 | 0.6 | 0.3 | 0 | 0.3 | 83 |
| Grey teal | 2.6 | 25.6 | 19.2 | 30.7^ | 74.0 | 0.3 | 9.0^ | 53.3 | 7.8 | 16.0 | 36.8 | 17.0 | 100 |
| Hardhead | 0 | 12.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 |

| Common name | Berry Jerry | Coonacoocabil | Dry Lake | Gooragool | McKennas | Molleys | Narrandera SF | Sunshower | Yarrada | Euroley (C1) | Turkey Flats (C2) | Yanco Ag (C2) | % Occurrence* |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Intermediate egret** | 0 | 0 | 0.5 | 5.4 | 7.0 | 1.2 | 0.5 | 2.4 | 1.6 | 0 | 0.2 | 0 | 67 |
| **Little black cormorant** | 1.0 | 5.0 | 1.5 | 21.6 | 18.0 | 0 | 0.5 | 9.6 | 1.2 | 16.8 | 0 | 0 | 75 |
| **Little egret** | 0 | 0 | 0 | 1.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 |
| **Little pied cormorant** | 17.0 | 1.6 | 0.4 | 11.5 | 28.8 | 5.5 | 0.2 | 4.7 | 4.8^ | 7.8 | 0.3 | 0 | 92 |
| **Masked lapwing** | 0 | 0 | 0.3 | 0 | 1.3 | 0 | 0 | 0 | 0 | 0 | 0 | 0.5 | 25 |
| **Rufous night heron** | 0 | 0 | 0 | 2.0^ | 0 | 0.3 | 5.5 | 0.3 | 0 | 0 | 0 | 0 | 33 |
| **Pacific black duck** | 3.4^ | 14.8 | 2.0 | 16.3 ^ | 5.5 | 1.0 | 1.8 | 17.3 | 1.8 | 8.5 | 16.7 | 4.2 | 100 |
| **Pied cormorant** | 0 | 0 | 1.5 | 0 | 0 | 0 | 0 | 0 | 0.5 | 0 | 0 | 0 | 17 |
| **Pink-eared duck** | 0 | 0 | 0 | 5.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 |
| **Plumed whistling duck** | 0 | 0 | 0 | 0 | 6.0 | 0 | 0 | 50.0 | 0 | 0 | 0 | 0 | 17 |
| **Purple swamphen** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 3.0 | 8 |
| **Red-kneed dotterel** | 0 | 0 | 7.0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 8 |
| **Royal spoonbill** | 0 | 0 | 2.3 | 2.0 | 8.3 | 0 | 0 | 2.7 | 1.0 | 0 | 3.0 | 0 | 50 |
| **Silver gull** | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1.0 | 0 | 8 |
| **Straw-necked ibis** | 0 | 0 | 0.2 | 6.6 | 2.6 | 4.6 | 0.8 | 0.2 | 0.8 | 0 | 0 | 0 | 58 |
| **White-faced heron** | 1.2 | 1.4 | 3.8 | 1.2 | 4.2 | 0.8 | 1.0 | 3.8 | 3.0 | 0.5 | 4.0 | 2.8 | 100 |
| **White-necked heron** | 0.3 | 0 | 0.3 | 2.6 | 0 | 3.0 | 0 | 1.6 | 7.3 | 0.2 | 3.4 | 0.4 | 75 |
| **Yellow-billed spoonbill** | 0 | 0.4 | 2.6 | 11.3 | 24.2 | 0 | 0.4 | 7.3 | 0.2 | 0.5 | 0.5 | 1.0 | 83 |
| **Total species** | 13 | 12 | 22 | 24 | 21 | 13 | 16 | 22 | 18 | 12 | 15 | 13 | |

*Occurrence represents the proportion of sites where each species was observed. ^ Species recorded nesting and/or with young. Status: J = JAMBA, C = CAMBA (International migratory bird agreements Australia has with Japan and China, respectively). Nomenclature taken from the Handbook of Australian, New Zealand and Antarctic Birds (Marchant and Higgins 1990, 1993; Higgins and Davies 1996).

# Appendix 5. Fish and frog species recorded in the mid-Murrumbidgee wetlands (2011–12)

Table 11. Fish and frog species recorded in the mid-Murrumbidgee wetlands (2011–12).

| Common name | Scientific name |
|---|---|
| | |
| ***Native fish*** | |
| **Australian smelt** | *Retropinnia semoni* |
| **Bony bream** | *Nematalosa erebi* |
| **Carp gudgeon** | *Hypseleotris spp.* |
| **Murray-Darling rainbow fish** | *Melanotaenia fluviatilis* |
| **Unspecked hardyhead** | *Craterocephalus stercusmuscarum fulvas* |
| **Introduced fish** | |
| **Gambusia (mosquito fish)** | *Gambusia holbrooki* |
| **European carp** | *Cyprinus carpio* |
| **Goldfish** | *Carassius auratus* |
| **Redfin perch** | *Perca fluviatilis* |
| **Oriental weatherloach** | *Misgurnus anguillicaudatus* |
| ***Frogs*** | |
| **Plains froglet** | *Crinia parinsignifera* |
| **Barking marsh frog** | *Limnodynastes fletcheri* |
| **Inland banjo frog** | *Limnodynastes interioris* |
| **Peron's tree frog** | *Litoria peronii* |
| **Spotted marsh frog** | *Limnodynastes tasmaniensis* |
| ***Other*** | |
| **Long-necked turtle** | *Chelodina longicollis* |
| **Macquarie river turtle** | *Emydura macquarii macquarii* |
| **Freshwater prawn/shrimp** | *Macrobrachium* spp. |
| **Yabby** | *Cherax destructor* |

# Appendix 6. List of Algal species sampled

Table 12. List of Algal species sampled in the Murrumbidgee, Tumut and Goobarragandra Rivers between June 2011 and October 2011.

| Group | *Species* |
|---|---|
| Rhodophyta (red algae) | *Audounella* |
| Chlorophyta (green algae) | *Closterium* |
| | *Cosmarium* |
| | *Draparnaldia* (large and small forms) |
| | *Oedogonium* (female) |
| | *Scenedesmus sp* |
| | *Stigeoclonium* |
| | *Ulothrix* |
| | *Spirogyra* |
| | *Kirchella* |
| Cyanophyta (blue green algae) | *Leptolyngbya* |
| | *Lyngbya* |
| | *Oscillatoria* |
| | *Phormidium* |
| | *Planktolyngbya* |
| | *Planktothrix* (fat and thin forms) |
| | *Pseudoanabaena* |
| | *Achnanthes cf.subexigua* |
| Bacillariophyta (diatoms) | *Achnanthidium minutissima* |
| | *Planothidium* |
| | *Aulocoseira granulate* |
| | *Cocconeis placentula* |
| | *Cyclotella* |
| | *Cymbella cistula* |
| | *Cymbella tumidia* |
| | *Encynema silesiacum* |
| | *Encynema minutum* |
| | *Encynopsis* |
| | *Fragillaria* (chain and large forms) |
| | *Fragillaria capucina* |
| | *Gomphonema lagenula* |
| | *Gomphonema parvulum* |
| | *Gomphonema acuminatium* |
| | *Gomphonema clavatum/truncatum* |
| | *Gomphonema psuedoguagr* |
| | *Gyrosigma* |
| | *Hantzschia* |
| | *Melosira varians* |
| | *Navicula erifuga* |
| | *Navicula gregaria* |
| | *Navicula pusilla* |
| | *Navicula schroeteni* |
| | *Navicula veneta* |
| | *Navicula viridula* |
| | *Nitzschia gracilis* |
| | *Nitzschia frustulum* |

| Group (continued) | *Species*(continued) |
|---|---|
| | *Nitzchia capitellata* |
| | *Nitzschia tubicola* |
| | *Nitzschia palea* |
| | *Rossithidium* |
| Bacillariophyta (diatoms) | *Surrirella* |
| | *Pinnularia subcapitata* |
| | *Synedra ulna* |
| | *Tabellaria flocculosa* |
| | *Tryblionella* |

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