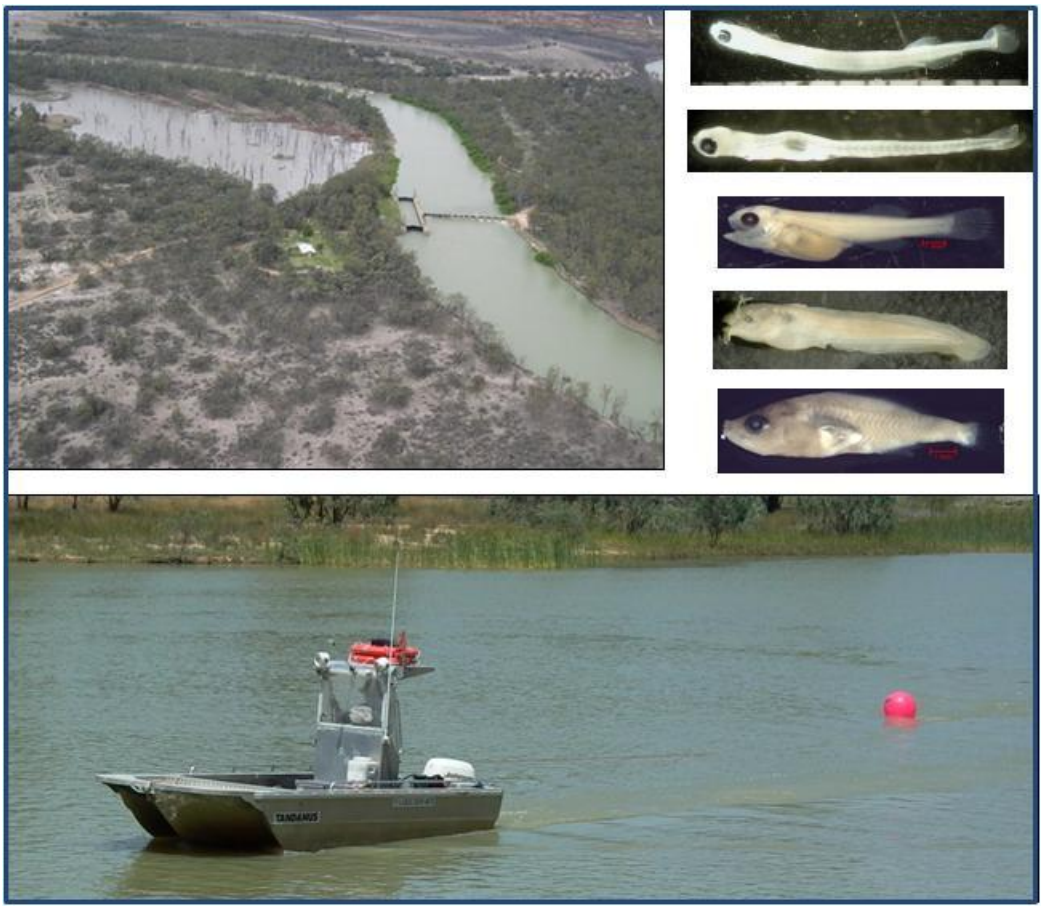
Australian Government

Commonwealth Environmental Water Office

Monitoring of ecological responses to the delivery of Commonwealth environmental water in the lower River Murray, during 2011-12

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



June 2015

Ye, Q.[1], Aldridge, K.[2], Bucater, L.[1], Bice, C.[1], Busch, B.[2], Cheshire, K.J.M[1], Fleer, D.[1], Hipsey, M.[3], Leigh, S.J.[1], Livore, J.P.[1],Nicol, J.[1], Wilson, P.J. [1] and Zampatti, B.P. [1]

(2015). Monitoring of ecological responses to the delivery of Commonwealth environmental water in the lower River Murray, during 2011-12. Final report prepared for Commonwealth Environmental Water Office. South Australian Research and Development Institute, Aquatic Sciences.

[1] SARDI Aquatics Sciences
[2] The University of Adelaide
[3] The University of Western Australia

This monitoring project was commissioned by the Commonwealth Environmental Water Office to the South Australian Research and Development Institute (SARDI), Aquatic Sciences. The component of 'salt and nutrient transport' monitoring and modelling was sub-contracted to the University of Adelaide and the University of Western Australia. The project was funded by the Commonwealth Environmental Water Office with in kind contribution from the South Australian Environmental Protection Authority (EPA) and the Department of Environment, Water and Natural Resources (DEWNR).

Copyright

This report should be attributed as 'Monitoring of ecological responses to the delivery of Commonwealth environmental water in the lower River Murray, during 2011-12, Commonwealth of Australia 2015'.

Cover Images:

Source of the photo of the lower River Murray: DEWNR

Source of the photos of larval fishes and research vessel Tandanus: SARDI

ACKNOWLEDGEMENTS

This study was funded by the Commonwealth Environmental Water Office (CEWO) in 2011-12 with in kind contribution from the Environment Protection Authority (EPA) South Australia.

For the salt and nutrient transport study, data for the Lower Lakes was provided by the South Australian Department of Environment, Water and Natural Resources (DEWNR) and EPA as part of the South Australian Government's $610 million *Murray Futures* program funded by the Australian Government's *Water for the Future* initiative. The model used has previously been developed and validated for Land and Water Australia, SA Water, DEWNR and EPA.

For the larval fish assemblage study, previous years' data (2005-06 to 2008-09 and 2010-11) were collected with funding provided through the Centre for Natural Resources Management, the Murray–Darling Basin Authority's (MDBA) The Living Murray Program, the South Australian Murray–Darling Basin Natural Resource Management Board, DEWNR and the Goyder Institute for Water Research.

Special thanks to Paul Searle and Tim Branford (DEWNR) for undertaking the ADCP transects and providing data. Thanks to Charles James (SARDI) for assistance with data collation and analysis for the hydraulic diversity study. Thanks to Ian Magraith, Arron Strawbridge, Neil Wellman, Thiago Vasques Mari, Hanru Wang and David Short (SARDI), for assistance in the field or laboratory work for the larval fish assemblage and golden perch recruitment studies. David Crook (Charles Darwin University) and Jed McDonald (SARDI) processed otoliths using LA-ICPMS and provided invaluable advice and assistance interpreting $^{87}Sr/^{86}Sr$ data. We also appreciate the assistance of SA Water staff from Locks 1, 6 and 9 in collecting water samples for Sr analysis. Thanks also to Ryan Breen (CEWO), Jarod Eaton (DEWNR) and John Waterworth (MDBA) for assistance in trying to decipher

environmental water delivery as a component of daily flow in the lower River Murray.

TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

EXECUTIVE SUMMARY

This project focuses on the intervention monitoring of the ecological responses to Commonwealth environmental water releases to the lower River Murray in 2011-12. The lower River Murray encompasses a wide range of aquatic habitats that support diverse species of native flora and fauna. This complex ecosystem is strongly influenced by variation in riverine flow regime. During the prolonged drought that affected the Murray–Darling Basin (2001–2010), the ecological community in the lower River Murray suffered severe stress. Since then, Commonwealth environmental water has been delivered alongside natural flows in order to contribute to the recovery of water-dependent ecosystems and build resilience against future stress.

During 2011-12, 329 gigalitres of Commonwealth environmental water was delivered to the lower River Murray to seek a number of ecological outcomes including enhancing spawning and recruitment of large-bodied native fish, such as golden perch (*Macquaria ambigua ambigua*). Environmental water delivery during December 2011 and January 2012 enhanced a within-channel natural flow pulse up to 26,000 megalitres per day (ML d^{-1})(at the South Australian border) and extended the flow recession during summer. The watering action supported enhanced hydrological variability in the lower River Murray and maintained flows above ~15,000 ML d^{-1} at the South Australian border during this period.

The current project investigated four key components of ecological responses during 2011-12 in relation to key objectives of environmental watering actions for the lower River Murray, including

(1) hydraulic diversity in the river channel,

(2) golden perch larvae and larval fish assemblage,

(3) golden perch recruitment and natal origin, and

(4) salt and nutrient transport.

Hypotheses were proposed based on our conceptual understanding of the life histories of relevant biota and ecological processes, and what responses might be expected from the flow scenarios and environmental water delivery in 2011-12. The four components are addressed as four studies. This report is a synthesis of these studies. More detailed information can be found in specific technical reports (Aldridge *et al.* 2013; Bice *et al.* 2013; Ye *et al.* 2013).

Key ecological outcomes

Monitoring in 2011-12 identified a number of ecological responses associated with the delivery of Commonwealth environmental water in the lower River Murray. Key outcomes are summarised in Table 1a. Reponses of large-bodied fish larvae in high flow years are also summarised in Table 1b.

Table 1a. Summary of key ecological outcomes associated with Commonwealth environmental water releases to the lower River Murray and associated watering during 2011-12. * Fish recruitment and natal origin studies are not included in this report.

Objective of the watering	Expected outcome	Indicator	Monitoring/modelling result
Hydraulic habitat conditions	Increased flows will lead to increased velocities and hydraulic habitat in the river channel	Hydraulic complexity (mean velocities and variability in velocities and circulation)	Greater hydraulic complexity was evident in the main channel at flow of 13,000–16,000 ML d^{-1} relative to previous assessments at flow of <4,000 ML d^{-1} A significant increase in hydraulic complexity with increased discharge (23,000–33,000 ML d^{-1})* * Whilst not a product of Commonwealth environmental water delivery this result informs future environmental watering
Reproduction of large-bodied fish (particularly golden	Increased flows will enhance reproduction of golden perch and silver perch in the South	Presence of early stage larval fish (golden perch and silver	An extended period with presence of golden perch larvae as a consequence of local spawning and/or larval drift from other spawning areas following the summer flow pulses supported by

Objective of the watering	Expected outcome	Indicator	Monitoring/modelling result
perch)	Australian reaches of the lower River Murray	perch) Larval fish assemblage structure (abundance and species composition)	environmental watering Larval fish assemblage structure changed due to increased abundances of golden perch and silver perch
As above	As above	Age of golden perch lavae and young-of-year (YOY) Spatial origin of golden perch lavae and YOY using otolith chemistry	Golden perch that recruited to the lower River Murray in 2012 were generally not spawned in association with the delivery of Commonwealth environmental water to the lower Murray channel. Nevertheless, substantial volumes (4,000–5,000 ML d^{-1}) of environmental water delivered down the Darling River from January–February 2012 would have aided the downstream dispersal of early life stage golden perch spawned in the Darling and the subsequent recruitment of these fish in the lower Murray
Salt and nutrient transport and export (water quality and nutrient cycling)	Increased flows will transport salt through the River Murray and export the salts from the system, providing sufficient flow out of the Murray Mouth Increased flows will transport nutrients through the River Murray, Lower Lakes and Coorong, stimulating primary productivity	Salt transport – Lock 1 to Southern Ocean Nutrient transport – Lock 1 to Southern Ocean By ELCOM-CAEDYM model	A significant reduction in salinity levels in the Northern Coorong. The greatest influence was observed during February 2012, with modelling suggesting that salinity was reduced by up to 70% in the Northern Coorong) Increased export of salt and particulate organic nutrients to the Southern Ocean associated with environmental water releases, which were associated with elevated outflows through the Murray Mouth and reduced seawater incursions. The greatest influence was observed during February 2012, with modelling suggesting that environmental water accounted for 70% of the total salt exports and 50% of the

Objective of the watering	Expected outcome	Indicator	Monitoring/modelling result
			particulate organic nutrient exports from the Murray Mouth

Table 1b. Summary of larval fish reponses to high flow years including 2011-12 watering year.

Objective of the watering	Expected outcome	Indicator	Monitoring/modelling result
Reproduction of large-bodied fish	Reproduction of large-bodied fish including flow-cued spawning (i.e. golden perch & silver perch) and circa annual spawning species (e.g. Murray cod & freshwater catfish) may be enhanced by greater flows.	Larval abundance of large-bodied species	Abundance of large-bodied species including golden perch, silver perch, Murray cod and freshwater catfish larvae during flow years (e.g. 2010-11 and 2011-12) was greater than in drought years (e.g. 2005-06 and 2008-09). The flow regime during the 2011-12 spring and summer supported a larval fish assemblage consisting of large-bodied and small to medium-bodied species, thus supporting diversity.

Key learning and management implications

Based on outcomes from the 2011-12 monitoring, and current knowledge of fish biology, flow related ecology and nutrient dynamics in the lower River Murray, Lower Lakes and Coorong, the following points should be considered with regard to environmental watering:

1. Environmental flow management has typically focussed upon temporal variability in flow volume and floodplain inundation whilst often overlooking within-channel processes such as spatio-temporal variability in stream hydraulics.

2. Hydraulic complexity is an integral part of the habitat diversity in riverine systems and often strongly influences the life history cycles of aquatic biota and affects biodiversity and community structure.
3. Restoration of hydraulic diversity to the main channel of the lower River Murray is likely to be important for restoring key ecological processes and functions, given prior to regulation it was a flowing system with complex hydraulic conditions over a range of spatial scales, even at low flows (Mallen-Cooper *et al.* 2011).
4. Environmental flows can be managed to improve hydraulic diversity which could contribute to diversity in fish assemblages and facilitate life-history processes such as spawning and recruitment.
5. This study developed a quantitative relationship between the level of hydraulic complexity and discharge for flows between 13,000 and 33,000 ML d^{-1} and will thus inform future environmental flow delivery of similar volumes.
6. There was an increase in hydraulic complexity in the main channel with increased discharge (23,000–33,000 ML d^{-1} at the South Australian border), suggesting that a greater diversity of hydraulic microhabitats may be provided at flows of this magnitude, potentially resulting in a greater ecological response.
7. Some level of hydraulic complexity was present even at 13,000–16,000 ML d^{-1} but virtually absent at flows <4,000 ML d^{-1} (Lock 4-5 reaches, Kilsby 2008), suggesting the benefit of environmental water releases even to maintain flow at >15,000 ML d^{-1}. Whilst flows of ~15,000 ML d^{-1} may initiate local spawning of flow-cued species (golden perch and silver perch, *Bidyanus bidyanus*), they may also facilitate larval drift/dispersion from upstream and enhance recruitment of golden perch to the lower River Murray (see Ye *et al.* 2008, Zampatti and Leigh 2013a).
8. Fish spawning and larval assemblage structure were strongly influenced by flow. This study supports the notion that the two flow-cued spawning species, golden perch and silver perch, will spawn

in conjunction with overbank flows and increased within-channel flows. The relative abundance of the annual spawning species Murray cod (*Maccullochella peelii)* and freshwater catfish (*Tandanus tandanus*) larvae was also greater during higher flow years. Consequently, environmental watering that promotes within-channel and overbank increases in flow will enhance the spawning and recruitment, and improve the resilience of native fish populations.

9. Larval and juvenile golden perch collected in the lower River Murray may be the progeny of spawning events that occur in the Murray and/or Darling rivers over an extended period from October–January. Larval golden perch were generally only collected in the lower River Murray when flows exceed ~10,000 ML d^{-1} and relative abundance increases at flows >20,000 ML d^{-1}, thus, flows of this magnitude could be targeted during the reproductive season (mid spring–summer) of flow cued spawning fishes (i.e. golden and silver perch).
10. Whilst overbank flows (floods) are ecologically important, within-channel flow management presents an opportunity to enhance populations of golden perch and other native fish.
11. Variability in flows contained within the river channel will support reproduction (spawning and recruitment) of a range of species from different flow guilds (e.g. large-bodied flow-cued spawning species and small to medium-bodied 'flow independent' species) and thus, can contribute to promoting a diverse fish community in the lower River Murray.
12. Based on insights provided by this study and knowledge of nutrient dynamics in the Lower Murray, the following points could be used to help guide future environmental water use:
    - Environmental watering during moderate-high flow periods (e.g. >40,000 ML d^{-1}) are likely to have greater impacts on salt and nutrient exports from the Murray Mouth than during low flow

periods (e.g. <10,000 ML d^{-1}) when the Murray Mouth outflows are small. In contrast, environmental watering during low flow periods is likely to have greater impacts on salt and nutrient concentrations than during moderate-high flow periods.

- Maximum exports of salt and nutrients from the Murray Mouth are likely to be achieved by delivering environmental water during periods of low oceanic water levels (summer). However, whilst this may have short-term benefits, reduced water delivery at other times is likely to increase the import of material from the Southern Ocean. In contrast, delivery of environmental water to the region at times of high oceanic water levels is likely to increase the exchange of water and associated nutrients and salt through the Coorong, rather than predominately through the Murray Mouth.
- Environmental watering which result in floodplain inundation will likely result in increased nutrient concentrations (mobilisation) and export. This may be achieved by moderate-large flows (e.g. >40,000 ML d^{-1}) that inundate previously dry floodplain and wetlands.
- Environmental watering during winter may result in limited assimilation of nutrients by biota (slower growth rates), whilst releases during summer could increase the risk of blackwater events and cyanobacterial blooms, depending on hydrological conditions and the degree of wetland connectivity. Environmental watering during spring is likely to minimise these risks, but also maximise the benefits of nutrient inputs (e.g. stimulate productivity to support larval survival).
- Multiple watering events in a single year could jointly target several objectives. For example, providing one event in spring to increase nutrient assimilation, followed by a subsequent event to export material from the system may provide multiple benefits.

Recommendations for future research and monitoring

1. Understanding biological responses to flows (both hydrology and hydraulics) and identifying ecologically important components of flow regimes are important to guide environmental flow management at appropriate spatial and temporal scales.
2. Long-term research and monitoring that builds on key long-term datasets will enable stronger inference and comparison through time and across different flow scenarios and regimes.
3. The knowledge of flow related ecology should be analysed in the context of environmental flow management, which will not only best inform the optimisation of flow delivery to maximise ecological outcomes but also help set sound ecological objectives/targets that could be realistically achieved with improved watering regimes.
4. Flow related ecological research could be conducted through monitoring biotic and environmental responses to natural flows or in conjunction with environmental water releases.
5. An improved understanding of how hydrodynamics vary with discharge will inform environmental flow delivery in regards to volumes likely to facilitate hydraulic complexity and biotic responses. Future monitoring could investigate water velocity data at flows of 4,000–13,000 ML d^{-1} to determine potential threshold flow rates beyond which hydraulic complexity begins to increase.
6. Research is required to understand the association of native fish species with hydraulic habitats (at biologically relevant scales) and the explicit link between hydraulics and ecological processes (e.g. spawning and recruitment).
7. Understanding the causal link between hydraulics and vital life-history processes, and determining the best hydraulic metrics to do so, will provide a tool to inform future environmental water delivery.
8. Environmental water requirements could be specified as the provision of particular hydraulic conditions and the flow, and

delivery strategies, required to create such conditions rather than a sole reliance on hydrological metrics.

9. Continued refinement of the ELCOM-CAEDYM model will further improve its capacity to assess the response of salt and nutrient dynamics to environmental watering in the hydrologically complex system of the Lower Murray. In future, it could also be used to assess various watering actions in both forecasting and retrospective analysis.

1 INTRODUCTION

River regulation and changes to the natural flow regimes have severely impacted riverine ecosystems throughout the world (Kingsford 2000; Bunn and Arthington 2002; Tockner and Stanford 2002). It is widely recognised that flow plays a pivotal role in maintaining the ecological integrity of riverine systems (Junk *et al.* 1989; Poff *et al.* 1997; Puckridge *et al.* 1998; Lytle and Poff 2004). Over the last decade, environmental flows have been increasingly applied worldwide for ecological restoration in river systems (Poff *et al.* 1997; Arthington *et al.* 2006). In Australia, over 2,400 GL of Commonwealth environmental water has been delivered, along with other environmental flows, since 2008-09 to protect or restore water dependent ecosystem condition in the Murray–Darling Basin (MDB) (www.environment.gov.au/ewater/). Understanding biological and ecological responses to flow variability is essential for environmental flow management to achieve the best ecological outcomes (Walker *et al.* 1995; Arthington *et al.* 2006).

1.1 Lower River Murray, hydrology and environmental watering in 2011-12

River regulation and water extraction have substantially altered the natural flow regimes in the MDB, leading to significantly reduced hydrological variability and increased water level stability (Maheshwari *et al.* 1995; Richter *et al.* 1996). The impact is most pronounced in the lower River Murray, where the installation of levies, five tidal barrages and 11 low level (<3 m) weirs and a high level of upstream extraction have changed a historically dynamic flowing river into a series of weir pools (Maheshwari *et al.* 1995; Walker 2006). The average annual flow discharge to the sea has declined by 61% (from 12,333 GL y^{-1} to 4,733 GL y^{-1}) and the probability of cease to flow through the Murray Mouth increased from 1% to 40% (CSIRO 2008). Such modification has had a profound impact on the ecological processes and communities in the

lower River Murray, Lakes and the Coorong (Walker 1985; Walker and Thoms 1993; Brookes *et al.* 2009).

From 2001 to 2010, the MDB experienced its most severe drought on record (Potter *et al.* 2010), during which flow discharge into the lower River Murray remained low (i.e. <15,000 ML d^{-1} at the South Australian border) and highly regulated (Figure 1). The drought was broken in late 2010 following a significant increase in flow and extensive flooding in the lower River Murray. The overbank flow was the largest discharge (at the South Australian border) since 1993, reaching a peak at 93,000 ML d^{-1} in February 2011. Discharge remained high in 2011-12 although it was substantially less than in the previous flood year, with water limited to within-channel flows.

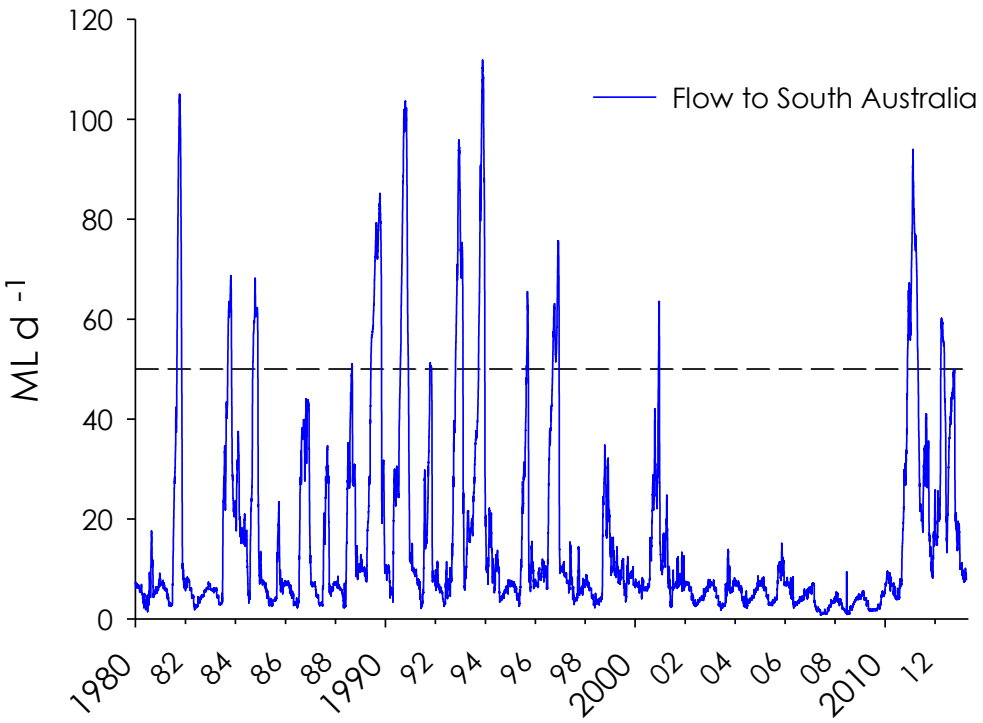


Figure 1. River Murray flow (ML d^{-1}) at the South Australian border from 1980–2012. Dashed line represents approximate bank-full flow in the lower River Murray.

During 2011-12, 329 GL of Commonwealth environmental water was delivered to the lower River Murray to complement natural river flows and provide additional freshes to the Lower Lakes and Coorong. This volume of environmental water was delivered from a combination of sources, specifically the Goulburn River and lower Broken Creek, the River Murray and lower Darling River, to achieve multiple benefits across the southern connected basin.

The environmental watering action in the lower River Murray aimed to enhance the magnitude and duration of natural flow pulses and reduce the rate of recession of natural flows. A number of expected outcomes were targeted by the watering action, including enhancing spawning and recruitment of large-bodied native fish, such as golden perch (*Macquaria ambigua ambigua*).

Environmental water use commenced during November 2011 following a period of high natural flows and continued to the end of January 2012. Environmental water use ceased on 3rd February 2012 due to high natural inflows from the Darling River. A large portion of environmental water was used between December 2011 and January 2012 to enhance a within-channel natural fresh up to 26,000 ML d^{-1}(at the South Australian border) and extended the flow recession during summer. The watering action contributed to improved hydrological variability in the lower River Murray and maintained flows above approximately 15,000 ML d^{-1} (at the South Australian border) during this period.

From February to June 2012, small volumes of additional environmental water were used to complement South Australian Entitlement Flow (Entitlement Flow is a perpetual and exclusive right to a seasonal allocation of a minimum of 3,000 ML d^{-1} during winter to 7,000 ML d^{-1} during summer). These volumes contributed to barrage releases into the Coorong and the transport of salt and nutrients from the River Murray system.

Throughout 2011-12, environmental water was also used to meet objectives in upstream catchments. Environmental water from other watering actions, including those in the Murrumbidgee River and mid-Murray region, contributed to sustaining flows in the lower River Murray and provided additional inflows to the Lower Lakes and Coorong (www.environment.gov.au/ewater/).

1.2 Intervention monitoring - background, aims and hypotheses for ecological components

This project focused on the intervention monitoring of key ecological responses during the delivery of environmental water to the lower River Murray (including the Lakes and Coorong) in 2011-12. The overall purpose of the monitoring program was to understand the ecological responses to environmental watering and contribute to the adaptive management of environmental flow regimes.

Aligning with relevant environmental watering objectives for 2011-12, the following four components of environmental and ecological outcomes were considered for the 2011-12 monitoring: (1) habitat, through hydraulic diversity in the river channel, (2) native fish reproduction through golden perch spawning and larval fish assemblage, (3) recruitment and natal origin of golden perch, and (4) water quality through salt and nutrient transport. These four components were investigated as distinct studies including the testing of hypotheses, which were proposed based on our understanding of the life histories of relevant biota and ecological processes, and what responses might be expected from the flow scenarios and environmental water delivery in 2011-12. The four ecological outcomes are addressed separately throughout this report.

1.2.1 Hydraulic diversity

Contemporary management and flow restoration in regulated rivers typically subscribes to the 'natural flow paradigm' (Poff *et al.* 1997), with the objective of restoring aspects of the natural flow regime.

Nonetheless, the delivery of environmental flows in the MDB has predominantly focused upon temporal variability in flow volumes and floodplain inundation. Hydrological variability also results in spatio-temporal variability in the physical characteristics of flow (i.e. hydraulics: depth, velocity and turbulence) within river channels, and the importance of such variability has, in the past, been an often overlooked objective of flow restoration. The hydraulic conditions experienced within a river are a function of discharge and physical features such as channel morphology, sediment type, woody debris and man-made structures (e.g. weirs), and hydraulic variability may directly influence river geomorphology and the diversity, distribution and abundance of aquatic biota (Statzner and Higler 1986; de Nooij *et al.* 2006).

Spatial complexity in flow hydraulics provides habitat heterogeneity across multiple scales. Several studies, both internationally and in Australia, have indicated the preference of different fish species, including salmonids, cyprinids, gobiids and galaxiids for 'patches' (i.e. micro-habitats) and 'reaches' (i.e. meso-habitats) with particular hydraulic conditions (Freeman and Grossman 1993; Henderson and Johnston 2010; Kilsby and Walker 2012). Thus, it stands to reason that a high diversity of hydraulic patches may result in high levels of biological diversity (de Nooij *et al.* 2006; Dyer and Thoms 2006). Additionally, the presence of microhabitats with particular hydraulic characteristics have been shown to influence the foraging efficiency of fish, including juveniles of some species (e.g. Atlantic salmon, *Salmo sala*), and is subsequently integral to fish retention in streams and potentially recruitment (Nislow *et al.* 1999). Furthermore, larval drift is an important process in the life cycle of many riverine fish species (e.g. golden perch) and the provision of particular hydraulic conditions is important in facilitating the downstream drift and dispersal of larval and juvenile fish (Brown and Armstrong 1985).

Prior to regulation, the lower River Murray was a flowing system, exhibiting heterogeneous hydraulic conditions over a range of spatial scales, even at low flows (Mallen-Cooper *et al.* 2011). Following regulation, the lower River Murray was transformed into a series of contiguous, predominantly still weir pools (Walker and Thoms 1993); variability in the flow regime was diminished and water levels are now relatively stable (Maheshwari *et al.* 1995, Blanch *et al.* 2000). Under low-volume regulated flows (<7,000 ML d^{-1}), which predominate, the lower River Murray is now hydraulically homogenous compared to unregulated reaches further upstream (Kilsby 2008) and this has likely had a profound impact on ecological character. Indeed, permanently flowing and hydraulically diverse habitats now only exist in selected off-channel anabranches of the lower River Murray (e.g. Chowilla Anabranch system) and subsequently, now represent the core habitat areas for species associated with flowing habitats (e.g. Murray cod) in the region (Zampatti *et al.* 2011).

Large floods in the lower Murray are little affected by river regulation, however, smaller floods and within-channel flow pulses are the component of the flow regime that has been most significantly altered by river regulation (Maheshwari *et al.* 1995). Paradoxically, flows of this magnitude (e.g. 15,000–50,000 ML d^{-1}) could practically be restored with the delivery of environmental water, within the current constraints of system operation. Hydraulic modelling indicates that as flows approach 15,000–20,000 ML d^{-1}, the river begins to regain its flowing nature (Mallen-Cooper *et al.* 2011), thus increasing hydraulic complexity and potentially promoting diversity in fish assemblages and facilitating life history processes such as spawning and recruitment. Nevertheless, changes in the hydraulic character of the river channel within this range of flows have not previously been verified in the field, yet these data are imperative to supporting the delivery of environmental water. Ultimately, the successful restoration of hydraulic

complexity will be paramount in restoring the ecological integrity of the lower River Murray.

The objective of this study was to investigate variability in hydraulic complexity in the lower River Murray between entitlement flows (3,000–7,000 ML d^{-1}) and flows of ~15,000 ML d^{-1}, resulting from environmental water delivery during 2011-12. Elevated natural flows were experienced over the study period meaning the original objective could not be met. Nevertheless, this provided the opportunity to investigate the same parameters at higher flow bands and inform future environmental water delivery at similar flow volumes. Specifically the aims were to 1) characterise hydraulic complexity (i.e. water velocities and circulation) under variable flow (i.e. 13,000–16,000 ML d^{-1} and 23,000–33,000 ML d^{-1}) by collecting real-time data and utilising various metrics, 2) use these data to 'ground truth' existing hydraulic models, and 3) integrate these data to inform the future delivery of environmental water in the lower River Murray in regards to flows required to reinstate hydraulic diversity.

1.2.2 Golden perch larvae and larval fish assemblage

River regulation and changes to the natural flow regime have a profound impact on ecosystem processes and aquatic biota, including fish populations (e.g. Gehrke *et al.* 1995; Freeman *et al.* 2001; Agostinho *et al.* 2004). Flow affects fish assemblages directly, by influencing critical life history processes including spawning, larval and juvenile survival, dispersion, movement/migration and subsequent recruitment (Welcomme 1985; Junk *et al.* 1989; Humphries *et al.* 1999; King *et al.* 2009). Fish assemblages are also affected by flow indirectly, by influencing floodplain inundation, productivity, channel morphology, hydraulic conditions, distribution of aquatic vegetation, diversity of structural elements, and habitat availability and selectivity (Nestler *et al.* 2012). In the MDB, there has been a significant decline in the abundance and distribution of native fish populations following

extensive modification of the river system and natural flow regimes (Walker 1985; Gehrke *et al.* 1995; Humphries *et al.* 2002; MDBC 2004).

To mitigate the adverse effects of river regulation on fish communities and aquatic food webs, environmental water has been used through manipulative flow releases to restore key elements of the natural hydrological regime that are deemed important for ecological processes and functions (Weisberg and Burton 1993; Travnichek *et al.* 1995; Cambray *et al.* 1997; Molles *et al.* 1998). These flow elements are also usually linked to the critical aspects of life history strategies of riverine fishes (Junk *et al.* 1989; Humphries *et al.* 1999; Lytle and Poff 2004). In Australia, environmental flows have been broadly applied as a 'restoration tool', particularly in the last five to ten years, and fish responses are often targeted among ecological outcomes with flows delivered to maintain critical habitats, improve connectivity, and/or facilitate spawning and recruitment (King *et al*. 2009; Watts *et al.* 2012). However, the lack of general relationships and good understanding of ecological responses to the various components of the flow regime sometimes restricts the ability to effectively manage environmental flows for specific rivers (Naiman *et al.* 2002; King *et al.* 2009; Poff and Zimmerman 2010).

In recent years, there has been some improvement of our knowledge of the role of flows and flooding in the life history cycles of many of the MDB fish (e.g. Humphries *et al.* 1999; King *et al.* 2003, 2009; Mallen-Cooper and Stuart 2003; Graham and Harris 2005; Ebner *et al.* 2009; Cheshire 2010; Cheshire *et al.* 2012). However, there are significant knowledge gaps on the specific environmental conditions required for successful fish spawning and recruitment and the complex mechanisms of how flow or other environmental factors affect fish (King *et al.* 2009). Such knowledge is critical for the restoration of ecologically important components of the flow regime in order to rehabilitate native fish populations through environmental flow management

(Walker *et al.* 1995; Arthington *et al.* 2006). Further eco-hydrological research is required, and investigations could be conducted through monitoring biotic responses to natural flows or in conjunction with environmental water releases.

This study focused on monitoring larval fish assemblage responses to the delivery of environmental water to the lower River Murray during 2011-12. Specific investigation was conducted on pre-flexion larvae of golden perch, a flow-cued spawning species, to provide insights on its spawning response to a summer within-channel flow pulse associated with the environmental watering during 2011-12. The data collected in the current year were compared with previous studies of larval fish assemblages (2005–2010) in the lower River Murray under different hydrological conditions (i.e. Cheshire and Ye 2008; Bucater *et al.* 2009; Cheshire *et al.* 2012). For the 2011-12 intervention monitoring, it was hypothesised that:

- following the summer within-channel flow pulse enhanced by the release of environmental water, golden perch larvae would be present in the lower River Murray for an extended period of time into late summer.
- increased flow with within-channel pulses would lead to changes in larval fish assemblage structure in the lower River Murray during 2011-12, compared to previous drought years (<10,000 ML d^{-1}), due to the presence and increased abundance of flow-cued spawning species (i.e. golden perch, silver perch) and a reduction in common small to medium-bodied species; larval assemblages in 2011-12 would be more similar to those in 2010-11.
- annual variation in larval fish assemblages would be correlated to changes in hydrology.

1.2.3 Recruitment and natal origin of golden perch

Restoring flow regimes to benefit aquatic ecosystems, including fish, requires an understanding of relationships between hydrology, life history and population dynamics (Arthington *et al.* 2006). Golden perch is one of only two native fish species in the River Murray, along with silver perch (*Bidyanus bidyanus*), considered to require increased discharge to initiate spawning (Humphries *et al.* 1999). Spawning and recruitment of golden perch in the River Murray corresponds with over bank flooding and increases in flow contained within the river channel (Mallen-Cooper and Stuart 2003; King *et al.* 2009; Zampatti and Leigh 2013a). In the lower River Murray (downstream of the Darling River junction) flows and water temperature nominally >15,000 ML d^{-1} and >20 °C, respectively, are considered to facilitate spawning and recruitment resulting in strong year-classes of golden perch (Zampatti and Leigh 2013b).

To understand the hydrological requirements of flow-cued spawning fish there is a need to be able to accurately determine the hydrological conditions at the time and place of spawning. This can be achieved by collecting fish in the act of spawning, collecting eggs, or determining the spatio-temporal provenance (i.e. when and where a fish was spawned) of early life stages (e.g. larvae), juveniles or adults.

Larval size/stage may be used as an indicator of daily age but age-at-size can be variable (Tonkin *et al.* 2006). To accurately determine the spawn date of golden perch, daily increments in otolith microstructure provide a validated method of determining daily age and hence back-calculating a spawn date (Brown and Wooden 2007).

Retrospectively determining where a fish was spawned can also be achieved by investigating otolith microstructure. The geochemical composition of otoliths reflects the chemistry of ambient waters at the time of deposition, although the uptake of some trace elements may be physiologically altered (Campana 1999). Elemental stable isotope

ratios, however, remain unaltered following incorporation into the otolith and directly reflect ratios in surrounding waters (Hobbs *et al*. 2005). Integration of otolith biochronolgy and geochemistry can potentially be used to retrospectively determine where a fish has been and when it was there, if locations have chemically distinct isotopic signatures (Gillanders 2005).

Following the unexpected occurrence of a strong cohort of golden perch in the lower River Murray spawned in 2009-10, a year when flow at the South Australian border did not exceed ~10,000 ML d^{-1}, Zampatti and Leigh (2013b), proposed a Darling River origin for these fish and that Strontium (Sr) isotopic ratios in otoliths could potentially be used to verify this hypothesis. Strontium isotope ratios in water are an artefact of catchment geology, and the Murray and Darling River systems have different Strontium isotope ($^{87}Sr/^{86}Sr$) signatures due to differences in the composition and age of rocks in their catchments (Douglas et al. 1995). Preliminary data indicate that the 2009-10 spawned cohort of golden perch originated in the Darling River and that otolith $^{87}Sr/^{86}Sr$ may prove a useful tool for determining the provenance of all life stages of golden perch (SARDI's unpublished data).

From July 2011 to June 2012 the CEWO delivered 329 GL of environmental water to the lower River Murray for a range of ecological objectives. The present study forms part of a broader monitoring program that aimed to assess the hydrological and ecological responses and benefits of this water and is concerned with the CEWO objective of stimulating and maintaining fish breeding and recruitment, particularly large-bodied native species, throughout the lower River Murray channel (CEWO 2012).

The aim of this study was to investigate golden perch recruitment in the lower River Murray in relation to flow, and determine the provenance of age 0+ recruits. We hypothesised that flows >15,000 ML d^{-1} during summer would result in recruitment (to age 0+) of golden perch in the

South Australian reaches of the River Murray, but that these young-of-year (YOY) fish would not necessarily have been spawned in the lower River Murray.

1.2.4 Salt and nutrient transport: Lock 1 to the Southern Ocean

Within aquatic ecosystems, flow provides habitat for aquatic organisms through the alteration of the physical and chemical environment. Salinity is an important parameter that governs the distribution and abundance of aquatic biota and is strongly influenced by flow (Williams 1987; Hart *et al.* 1991; Nielsen *et al.* 2003). Flow also results in the mobilisation and transport of nutrients, which allow organisms to grow (Poff *et al.* 1997). Nitrogen, phosphorus and silica are important nutrients because they often control the productivity of aquatic biota. These nutrients occur in various inorganic, organic, particulate and dissolved forms. For the purpose of this study we focused on:

- dissolved inorganic nutrients, including phosphate, ammonium (nitrogen) and silica, which are considered readily available to primary producers; and
- particulate organic phosphorus and nitrogen, which are associated with the biomass of living and dead (detritus) organisms.

Altering the flow regime of riverine systems has had significant consequences for salt and nutrient concentrations and transport. For example, extended periods of low flow can result in the accumulation of salts within aquatic ecosystems. In addition, extended periods of low flow with a lack of floodplain inundation can result in productivity being limited by the low availability of nutrients. Such observations have been made in the lower Murray, including the lower River Murray, Lower Lakes and Coorong (Brookes *et al.* 2009; Aldridge *et al.* 2011, 2012; Mosley *et al.* 2012). Environmental flow provisions may be used to reinstate some of the natural processes that control salt and nutrient concentrations and transport. In doing so, these flows provide

ecological benefits through the provision of habitat and resources to allow the growth and survival of biota. This study aimed to assess the changes in the salt and nutrient concentrations and transport in the lower River Murray (below Lock 1), Lower Lakes and Northern Coorong (Figure 2) associated with the delivery of Commonwealth environmental water between November 2011 and June 2012.

2 METHODS

2.1 Study sites

The lower River Murray in South Australia has no significant tributaries and its hydrology is determined by flows from the mid- and upper-Murray and Darling Rivers. The lower River Murray between the South Australian border and Wellington encompasses three distinct geo-morphological zones including the floodplain, lowland gorge and swampland regions (Figure 2), each with distinct ecological features. The River Murray discharges to the terminal lake system (Lakes Alexandrina and Lake Albert) before flowing to the Coorong estuary and Southern Ocean.

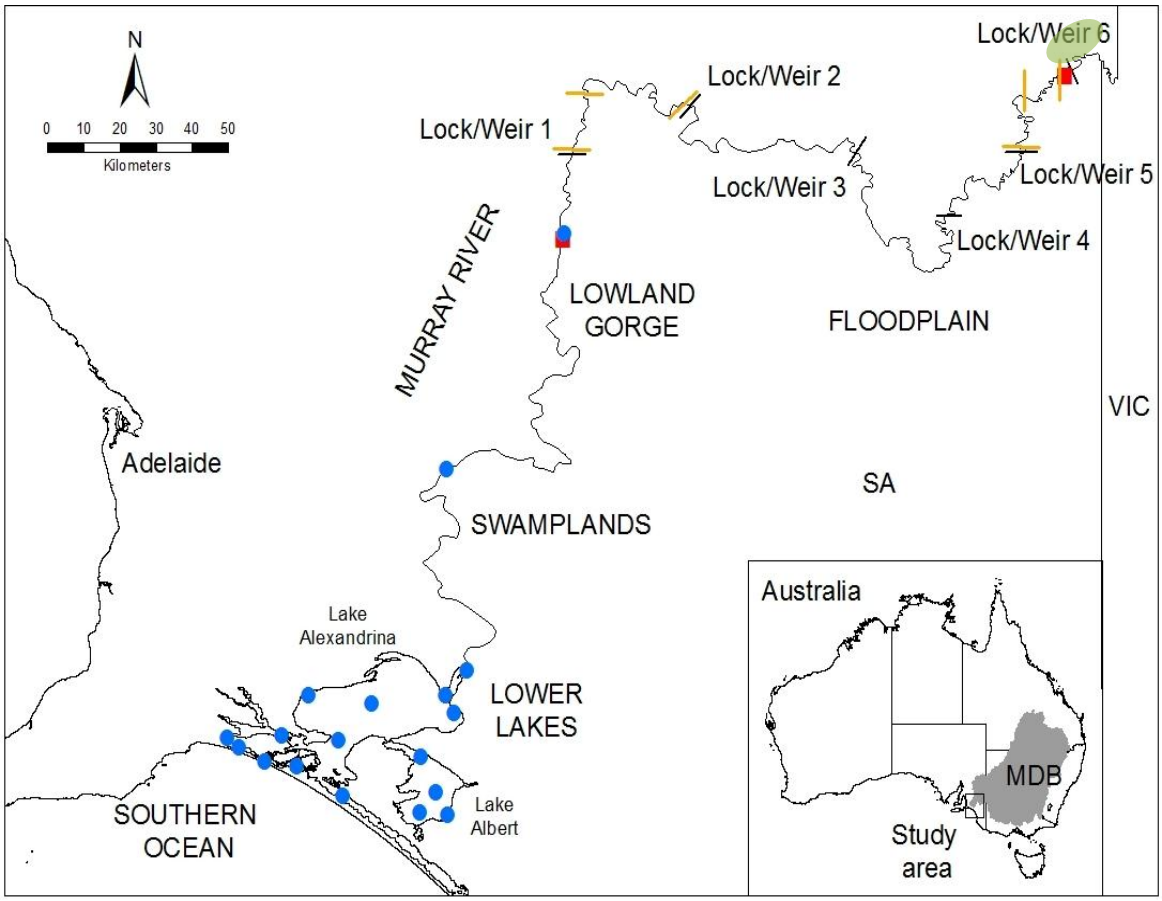


Figure 2. Study sites for hydraulic diversityyellow lines, larval fish assemblages, recruitment and natal origin of golden perch (red square) and salt and nutrient transport (blue circles) in the lower River Murray, Lower Lakes and Coorong in South Australia. Blue dots – water collection sites for salt and nutrient transport; red squares – larval fish sampling sites; yellow lines – hydraulic diversity sites and green shaded area – Chowilla Anabranch system.

2.1.1 Hydraulic diversity

Velocity profiles were measured across transects in two reaches (weir pools) in the main channel of the lower River Murray, one in the floodplain region, between Lock 5 and Lock 6 and the other in the gorge region, between Lock 1 and Lock 2 (Figure 2). Within each weir pool, cross-sectional velocity profiles were generated for three locations, 1) the upper weir pool (in the vicinity of the upstream weir), 2) the mid weir pool (approximately mid-way between the two weirs) and 3) the lower weir pool (within the vicinity of the downstream weir) (Figure 2).

2.1.2 Golden perch larvae and larval fish assemblages

Larval fish sampling occurred in the main channel of the lower River Murray in South Australia (Figure 2). Sampling was conducted at two sites in the tailwaters approximately 5 km downstream of Lock 1 (i.e. Site 1) and Lock 6 (i.e. Site 2) (Figure 2). The area surrounding Lock 1 is the gorge region, and the area surrounding Lock 6 is the floodplain region. Despite the surrounding floodplains being different, the main channel habitat is generally similar with wide, deep, slow flowing pool habitats.

2.1.3 Recruitment and natal origin of golden perch

To investigate the association between flow, golden perch recruitment (to at least YOY, age 0+) and natal origin we surveyed a total of 105 sites in the main channel of the lower River Murray in South Australia, the Chowilla Anabranch system and the littoral zones of Lake Alexandrina (Figure 2). Sites were sampled in the three distinct geomorphic regions of the lower River Murray (Walker and Thoms 1993): 1) swamplands and lakes (downstream of Mannum) (n = 29), 2) gorge (Mannum–Lock 3) (n = 29), and 3) floodplain (Lock 3–Lock 6) (n = 47) (Figure 2).

2.1.4 Salt and nutrient transport

This study considers the area between Lock 1 and the Southern Ocean. Water quality was assessed at numerous sites in the lower River Murray (below Lock 1), Lower Lakes (Lake Albert and Lake Alexandrina) and Northern Coorong (Figure 2). Data from the Lower Lakes were provided by the South Australian Environment Protection Authority through the Murray Futures funding program.

2.2 Hydraulic diversity

2.2.1 Survey technique

Cross-sectional velocity profiles were collected by DEWNR using a vessel mounted SonTek River Surveyor M9 Acoustic Doppler Current Profiler (ADCP). For specific details on the operation of ADCP units see Shields and Rigby (2005). Velocities were measured at two different flow bands, firstly during January 2012 at flows to SA (calculated River Murray discharge entering SA, i.e. QSA) of 16,214–16,898 ML d^{-1} and in March 2012 during flows of 33,241–38,627 ML d^{-1}. Velocities could not be measured during typical summer/autumn entitlement flows (3,000–7,000 ML d^{-1}) due to persistent high flows over the study period.

2.2.2 Data analysis

Data that were generated from ADCP transects were exported to MATLAB (The Mathworks Inc. 2010) and interpolated across grids with equal cell sizes (0.5 m long x 0.25 m high) using the Delaney triangulation scattered data function. Water velocities for each cell were generated in three planes; perpendicular or cross-transect (i.e. upstream to downstream), parallel to or along a transect (i.e. from bank to bank) and vertical velocity (i.e. up or down).

Analysis of variance (ANOVA) was used to determine significant differences in mean cross-transect velocities (U; Figure 3) between velocity profiles from the different locations (upper, mid and lower) and flow bands within each reach. Variability in U within cross-sections was

investigated by determining velocity ranges, standard deviation and the coefficient of variation. Mean velocities in the other planes, horizontally along transect (i.e. from bank to bank, W) and vertically (i.e. up or down, V) were also determined.

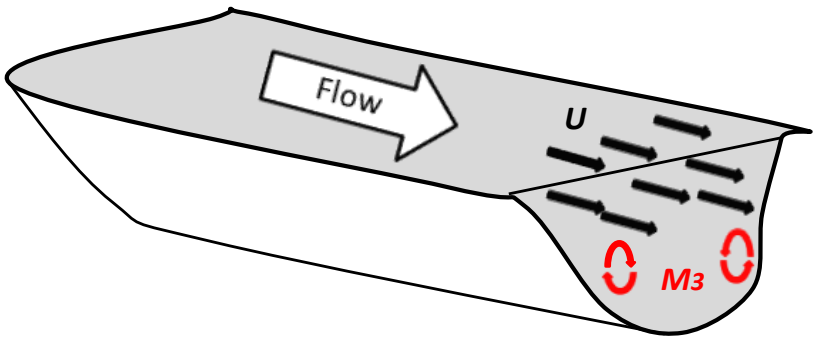


Figure 3. Schematic representation of a river reach and the hydraulic metrics investigated (after Shields and Rigby 2005), including variation in cross transect velocities (U) between transects and the modified circulation metric (M_3), which represents the area averaged frequency and strength of eddies or flow rotation within a cross-section.

We also adopt a spatial hydraulic metric developed by Crowder and Diplas (2000) known as the modified circulation metric to quantify flow complexity over a defined area, in this case, river cross-sections as measured during ADCP transects. The modified circulation metric (M_3; after Shields and Rigby 2005) represents a weighted average of flow rotation in the vertical plane per unit area (Figure 3). Calculation of M_3 is explained by Equation 1, where w represents velocity in the vertical plane z and v represents velocity in the lateral plane y. Higher values of M_3 indicate greater frequency and strength of eddies or greater levels of circulation (i.e. flow rotation) within a cross-section.

Equation 1 $$M_3 = \frac{\sum\left|\left(\frac{\Delta w}{\Delta y}-\frac{\Delta v}{\Delta z}\right)\right| * \Delta y * \Delta z}{\sum \Delta y * \Delta z}$$

2.2.3 Ground-truthing existing hydraulic models

'Measured' mean velocities were compared with 'modelled' mean velocities for the same sites using two modelling approaches. Firstly, hydrodynamic modelling has been undertaken in the Lock 6 region by Water Technology as part of a project investigating the risks posed to fish by the operation of the Chowilla regulator (Mallen-Cooper *et al.* 2011) utilising a one-dimensional Mike11 model. Mean velocities were modelled at flows (QSA) of 15,000 , 20,000 and 40,000 ML d^{-1}, at the Lock 5–6 upper weir pool site and are compared against velocities measured in the current study. The second set of hydrodynamic modelling was undertaken by DEWNR using MIKE FLOOD software (DEWNR unpublished). This model generated mean water velocity outputs for the lower, middle and upper weir pool sites of the Lock 1–2 and Lock 5–6 reaches at flows (QSA) of 13,000 ML d^{-1} and 24,000 ML d^{-1}

2.3 Golden perch larvae and larval fish assemblage

2.3.1 Sampling regime, larvae collection and processing

Larval fish sampling was conducted at two sites using plankton tows approximately fortnightly between October and March, covering the period of environmental water delivery in the 2011-12 water year. This method is consistent with previous larval fish studies in the lower River Murray (Cheshire and Ye 2008; Cheshire *et al.* 2012), with the difference being that the sampling period was extended to measure response to environmental flow release. Each site was sampled with three replications during the day-time and at night-time, of the same day, and both sites were sampled within a three-day period. Day-time and night samples were taken to gain a representative picture of the whole larval assemblage (Cheshire 2010).

Plankton tows were conducted using a set of paired square-framed bongo nets with 500 µm mesh; each net was 0.5 x 0.5 m and 3 m long. Nets were equipped with a pneumatic float in the centre of the frame, which allowed the frame to sit ~15 cm below the water surface. The

net was towed in a circle, astern for 15-minute intervals using a 20 m rope. The volume of water (m^3) filtered through each net was determined using a calibrated flow meter placed in the centre of the mouth openings.

Samples were preserved in 95% ethanol *in situ* and returned to the laboratory for sorting. All larvae were identified to species level where possible, with the aid of published descriptions, with the exception of carp gudgeon (*Hypseleotris spp*.), hardyhead (*Craterocephalus spp.*) and flatheaded gudgeon (*Philypnodon spp.*). Each of these three genera was treated as a species complex due to their close phylogenetic relationships and very similar morphologies making clear identifications difficult (Bertozzi *et al.* 2000; Serafini and Humphries 2004). In 2010-11, newly hatched larvae (free embryos) were collected, which were confirmed to be either golden perch or silver perch; however, due to very similar morphologies, they were subsequently grouped as 'perch free embryos' for data analysis. In both 2010-11 and 2011-12, at least two different types of eggs were sampled (likely to be Australian smelt, *Retropinna semoni*, and perch eggs); however, due to uncertainty, these were not included in data analysis.

Total length (TL) measurements of golden perch larvae collected in 2011-12 were taken using a dissecting microscope with different magnifications (x 6.5 – 40) depending on the size of the larvae (Figure 4). Since golden perch flexion occurs between 4.9–7.3 mm TL (Serafini and Humphries 2004), the mean 6.1 mm TL was used as a cut-off for the classification of larvae as pre-flexion in this study.

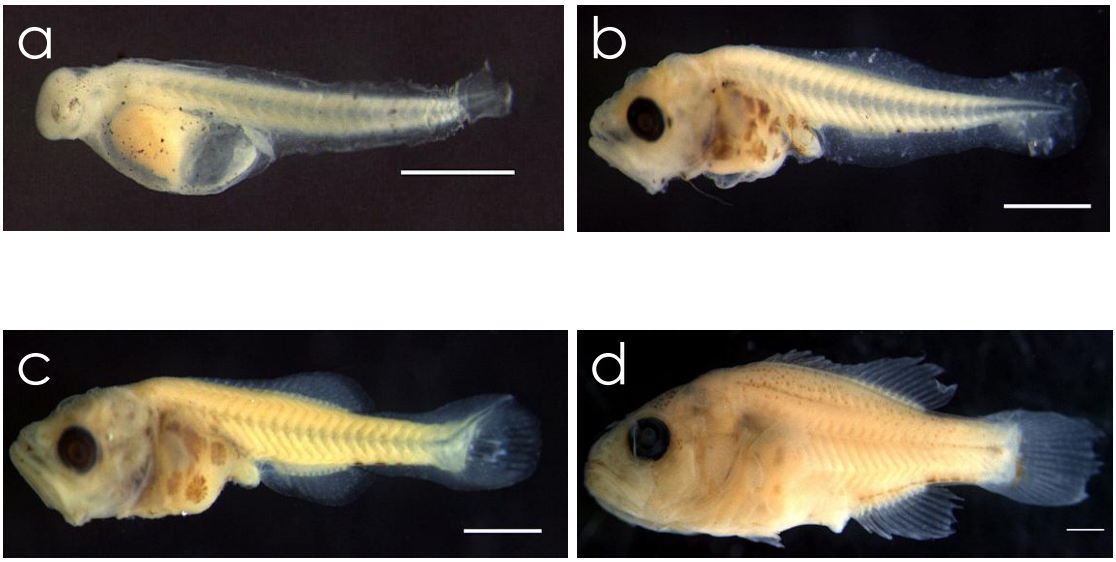


Figure 4. Developmental stages of golden perch larvae: a) and b) pre-flexion larvae; c) and d) post-flexion larvae. Scale bar 1 mm.

2.3.2 Environmental variables and data analysis

Data for flow (discharge in ML d^{-1}), water level (metres AHD, level relative to Australian Height Datum), water temperature (°C), and conductivity (µS.cm^{-1} @ 25 °C) were obtained for both sites, throughout the study, from the DEWNR Surface Water Archive (www.waterconnect.sa.gov.au; accessed on 10/08/2012). Water level data were transformed to relative water levels by measuring their deviation from 'normal' pool height (Site 1: 0.75 m, Site 2: 16.3 m).

The volume of water filtered through each plankton net was used to calculate the standardised abundance of fish per m^3. Six plankton tows generated six replicates for each site during each sampling trip. To account for the dispersion or concentration of larval fish under the extremely variable flow conditions, the standardised abundance of fish per m^3 was multiplied by the discharge volume at the time of sampling day to provide the relative abundance, as a flow corrected abundance indicator which was used for all analyses. Fish assemblage structure was characterised by the species composition and the relative abundance of each species.

Statistical analysis was conducted on the relative abundance of pre-flexion golden perch larvae to detect the difference between years (2011-12 *vs* previous years) and two sampling sites. Only samples from October to mid January were included in the statistical analysis because larval sampling was not conducted from late January to March in years prior to 2011-12. The temporal change in relative abundance of pre-flexion golden perch larvae during spring and summer was plotted against the flow discharge to South Australia for each year. Multivariate statistical analyses were undertaken on larval fish assemblages to investigate the difference in assemblage structure between 2011-12 and previous years. Detailed statistical analyses are described in Ye *et al.* (2013).

2.4 Recruitment and natal origin of golden perch

2.4.1 Hydrology

Daily mean flow (ML d^{-1}) and water temperature (°C) during the period July 2011 to July 2012 for the River Murray at the South Australian border, Euston, Frenchman's Creek (Lake Victoria inlet), Rufus River (Lake Victoria outlet) and the Darling River at Burtundy (Appendix I) were obtained from the South Australian Department of Environment, Water and Natural Resources (DEWNR, unpublished data) and the Murray–Darling Basin Authority (MDBA).

2.4.2 Water samples

To determine temporal variation in water $^{87}Sr/^{86}Sr$ over the period of environmental water delivery to the lower Murray, water samples were collected fortnightly from 19 January 2012 to 18 April 2012 at Locks 1, 6 and 9. One-off samples were also collected from the Darling River, River Murray (upstream and downstream of the Goulburn River), and Goulburn and Murrumbidgee Rivers (Appendix I).

2.4.3 Fish sampling

Adult and juvenile golden perch were sampled primarily by vessel electrofishing using a 7.5 kW Smith Root Model GPP 7.5 electrofishing unit. Sites throughout the main channel of the lower River Murray in South Australia were sampled in autumn (March/April) and spring (October/November) 2012 to maximise the chance of collecting YOY golden perch spawned in the spring–summer 2011-12 spawning season. Electrofishing was conducted during daylight hours and all available littoral habitats were fished. All fish were measured to the nearest mm (total length, TL) and a subsample (*n* = 7–104 fish) proportionally representing the length-frequency of golden perch collected from each geomorphic region was retained for ageing. Additional juvenile fish were obtained from plankton tows conducted as part of the larval fish sampling component (Section 2.3) in December–February 2012 and from *ad hoc* fyke net sampling in the lower lakes of the River Murray (swamplands and lakes geomorphic region) in early November 2012.

2.4.4 Ageing

Annual ageing

Golden perch exhibit considerable variation in length-at-age in the MDB (Anderson *et al.* 1992). Therefore, to accurately assess the age structure and year-class strength of golden perch, we investigated both length and age-frequency distributions. Fish retained for ageing were euthanized using AQUI-S® (Aqui-s, Lower Hutt, New Zealand) and sagittal otoliths were removed. Whole sagittae were embedded in clear casting resin and a single 400 to 600 μm transverse section was prepared. Sections of sagittae were examined using a dissecting microscope (x 25) under transmitted light. Estimates of age were determined independently by three readers by counting the number of discernible opaque zones (annuli) from the primordium to the otolith

edge. YOY (< 1 year old) fish were defined as individuals lacking clearly discernible annuli.

Daily ageing

To estimate the spawn date of larval and juvenile golden perch, daily variation in otolith microstructure was examined in a subsample of fish (*n* = 15). Golden perch larvae/juveniles were measured to the nearest mm and sagittal otoliths were removed. Transverse sections were prepared by embedding sagittae in Crystal Bond™ and grinding with 9 µm imperial lapping film from both anterior and posterior surfaces to the primordium. Sagittae were then polished using 0.3 µm alumina slurry to produce sections between 50 and 100 µm thick.

Sections were examined using a compound microscope (x 600) fitted with a digital camera and Optimas image analysis software (version 6.5, Media Cybernetics, Maryland, USA). Increments were counted blind with respect to fish length and capture date. Estimates of age were determined by counting the number of increments from the primordium to the otolith edge. Three successive counts were made by one reader for one otolith from each fish. If these differed by more than 5% the otolith was rejected, but if not the mean was used as an estimate of the number of increments. Increment counts were considered to represent true age of larval and juvenile golden perch and spawn dates were determined by subtracting the estimated age from the capture date (Zampatti and Leigh 2013b).

2.4.5 Water $^{87}Sr/^{86}Sr$ analysis

Twenty ml aliquots of each water sample were filtered through a 0.2 µm Acrodisc syringe-mounted filter into a clean polystyrene beaker and dried overnight in a HEPA-filtered fume cupboard. Previous analyses have shown that filtering after transfer to the laboratory, rather than after sample collection in the field, has no influence on measurement of $^{87}Sr/^{86}Sr$ (e.g. Palmer and Edmond 1989).

Strontium was extracted using a single pass over 0.15 ml (4 x 12 mm) beds of EICHROMTM Sr resin (50-100 µm). Following Pin *et al.* (1994), matrix elements were washed off the resin with 2 M and 7 M nitric acid, followed by elution of clean Sr in 0.05 M nitric acid. The total blank, including syringe-filtering, is ≤0.1 ng, implying sample to blank ratios of ≥4000; no blank corrections were therefore deemed necessary. Strontium isotope analyses were carried out on a "Nu Plasma" multi-collector ICPMS (Nu Instruments, Wrexham, UK) interfaced with an ARIDUS desolvating nebulizer, operated at an uptake rate of ~40 µL min^{-1}. Mass bias was corrected by normalizing to $^{88}Sr/^{86}Sr$ = 8.37521 and results reported relative to a value of 0.710230 for the SRM987 Sr isotope standard. Internal precisions (2SE) based on at least 30 ten-second integrations averaged ± 0.00002 and average reproducibility (2SD) was ± 0.00004.

2.4.6 Otolith preparation and $^{87}Sr/^{86}Sr$ analysis

Sagittal otoliths were dissected and mounted individually in Crystalbond™, proximal surface downwards, on an acid-washed glass slide and polished down to the primordium using a graded series of wetted lapping films (9, 5, 3 µm) and alumina slurry (0.5 µm). The slide was then reheated and the polished otolith transferred to a 'master' slide, on which otoliths from all collection sites were combined and arranged randomly to remove any systematic bias during analysis. The samples were rinsed in Milli-Q water (Millipore) and air dried overnight in a class 100 laminar flow cabinet at room temperature.

Laser ablation – inductively coupled plasma mass spectrometry (LA-ICPMS) was used to measure $^{87}Sr/^{86}Sr$ in the otoliths. The experimental system consisted of a ''Nu Plasma'' multi-collector LA-ICPMS (Nu Instruments, Wrexham, UK), coupled to a HelEx laser ablation system (Laurin Technic, Canberra, Australia, and the Australian National University) constructed around a Compex 110 excimer laser (Lambda Physik, Gottingen, Germany) operating at 193 nm. Otolith mounts were

placed in the sample cell and the primordium of each otolith was located visually with a 400× objective and video imaging system. The intended ablation path on each sample was then digitally plotted using GeoStar v6.14 software (Resonetics, USA). Each otolith was ablated along a transect from the primordium to the dorsal margin at the widest radius using a 6 ×100 µm rectangular laser slit. The laser was operated at 90 mJ, pulsed at 10 Hz and scanned at 3 µm sec^{-1} across the sample. Ablation was performed under pure He to minimise the re-deposition of ablated material, and the sample was then rapidly entrained into the Ar carrier gas flow. A pre-ablation step using reduced energy (50 mJ) was conducted along each transect to remove any surface contaminants and a 20-30 sec background was measured prior to acquiring data for each sample. Corrections for Kr and 87Rb interferences were made following closely the procedures of Woodhead *et al*. (2005) and mass bias was then corrected by reference to an $^{86}Sr/^{88}Sr$ ratio of 0.1194. Iolite Version 2.13 (Paton *et al*. 2011) that operates within IGOR Pro Version 6.2.2.2 (WaveMetrics, Inc., Oregon) was used to process data offline, with data corrected for potential Ca argide/dimer interferences.

A modern marine carbonate standard composed of mollusc shells ($^{87}Sr/^{86}Sr$ value of 0.70916 according to long-term laboratory measurements, identical to the accepted modern seawater value of 0.709160 - MacArthur and Howarth 2004) was analysed after every 10 otolith samples to allow for calculation of external precision. Mean (±1 SD) values of $^{87}Sr/^{86}Sr$ values in the modern marine carbonate standard (n = 24) run throughout the analyses were 0.70918 ± 0.00017, with external precision (expressed as ± 2 SE) calculated as ± 0.00006. Mean within-run precision, measured as ± 2 SE, was ± 0.00005.

2.5 Salt and nutrient transport

Water quality was assessed between November 2011 and June 2012 (Table 2). At each sampling site, measurements of water temperature,

electrical conductivity, dissolved oxygen, pH and turbidity were taken. In addition, water samples were collected and sent to the Australian Water Quality Centre, an accredited laboratory of the National Association of Testing Authorities, for analysis of nutrient concentrations.

Table 2. Sampling dates of sites in the three areas of study of salt and nutrient transport.

Sites	Sampling dates
Lower River Murray	01/02/12, 16/02/12, 29/02/12, 15/03/12, 29/03/12, 17/04/12, 3/05/12, 17/05/12, 8/06/12, 28/06/12
Lower Lakes	15/11/11, 13/12/11, 12/01/12, 16/02/12, 14/03/12, 12/04/2012, 15/05/12, 27/06/12
Coorong	02/02/12, 15/02/12, 28/02/12, 14/03/12, 28/03/12, 16/04/12, 2/05/12, 16/05/12, 7/06/12, 27/06/12

The physico-chemical information was incorporated into a three-dimensional hydrodynamic-biogeochemical model, ELCOM-CAEDYM, which has been used extensively within the region (Hipsey and Busch 2012). For this study, two simulations were run and compared for 1 October 2011 to 1 July 2012 – with and without environmental water provisions. The model was initialised with data from a range of sources, including Lock 1 inflows with and without environmental water provisions (provided by the MDBA). The flow data were treated as indicative due to complexities around interstate water accounting, which are likely to have resulted in an underestimation of environmental flow volumes and thus an underestimation of the influence of environmental flows on salt and nutrient transport.

Although assumptions such as these result in uncertainty in the model outputs (refer to Aldridge *et al.* 2013 for more detail), the model proved to be useful in assessing the general water quality response. The differences in salt and nutrient concentrations between the three water-bodies studied were captured in the model outputs. However, temporal differences at individual sites were not captured particularly

well in the model outputs. When assessing the relative differences between the two scenarios the uncertainties influence the accuracy of both equally. Consequently, modelled concentrations and loads should not be treated as absolute values, but instead as an indication of the general response to environmental water provisions in the region. For detailed information on the modelling approach refer to Aldridge *et al.* (2013) and Hipsey and Busch (2012).

3 RESULTS

3.1 Hydraulic diversity

3.1.1 Hydrograph

Flow within both river reaches varied over the period November 2011–April 2012 (Figure 5a and b). In the Lock 5–6, reach the upper weir pool transect is upstream of the Chowilla Creek junction, whilst the mid and lower weir pool transects are below the junction. As such, flow at Lock 6 is most relevant to the upper weir pool site whilst flow to SA (QSA) is most relevant to the mid and lower weir pool transects. During data collection in January 2012, flow at Lock 6 was 13,343 ML d^{-1} and QSA was 16,214 ML d^{-1}, increasing in March to 25,815 and 33,241 ML d^{-1} respectively (Figure 5a). Flow at Lock 1 (as a proxy for the Lock 1–2 reach) was 13,725 ML d^{-1} during data collection in January and 23,712 ML d^{-1} in March (Figure 5b).

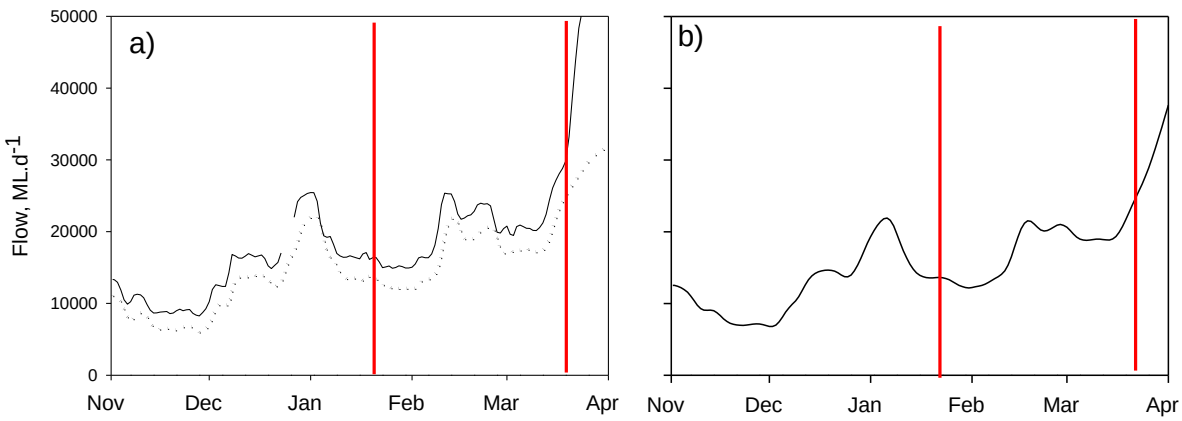


Figure 5. Daily river flow (ML d^{-1}) at a) Lock 6 (dotted line) and QSA (solid line) and at b) Lock 1 (as a proxy for the Lock 1–2 reach), from November 2011–April 2012. Red lines represent dates and flows when ADCP transects were conducted.

3.1.2 Hydraulic complexity

Table 3 presents a range of hydraulic metrics that characterised the hydraulic environment at each site during transect measurement in January and March 2012. Figure 6 presents an example of ADCP cross-sectional velocity profiles (Figure 6a, c) and circulation profiles (Figure

6b, d) from the Lock 1–2 upper weir pool location generated in January and March 2012. In all cross-sections, the lowest velocities were recorded near the edges of transects, with more uniform and higher velocities in the middle of the river and often higher in the water column (Figure 6).

Table 3. Hydraulic habitat metrics calculated from ADCP generated data from the upper, mid and lower weirpool locations within the Lock 1–2 and Lock 5–6 reaches in January and March 2012. Metrics include point discharge ($m^3.s^{-1}$) at each location, the transect length (m), mean depth (m) across the cross-section, total area of the cross-section (m^2), mean cross-transect (upstream to downstream) velocity (U, $m.s^{-1}$), range of cross-transect velocities (U range), standard deviation in cross-transect velocities ($m.s^{-1}$), mean velocity along or parallel to each transect (V, $m.s^{-1}$), mean velocity in the vertical plane (W, $m.s^{-1}$) and the modified circulation metric (M_3, s^{-1}).

	January			March		
			Lock 1–2			
	Upper	**Mid**	**Lower**	**Upper**	**Mid**	**Lower**
Discharge ($m^3.s^{-1}$)	195.98	185.56	182.36	360.68	353.26	324.92
Transect length (m)	112	109	157	122.35	113.18	167.02
Mean depth (m)	1.91	2.64	2.65	2.08	2.70	2.71
Area (m^2)	528.63	650.26	954.66	584.18	698.71	980.22
Mean *U* ($m.s^{-1}$)	0.37	0.29	0.19	0.62	0.51	0.33
U range ($m.s^{-1}$)	-0.08–0.78	-0.16–0.66	-0.17–0.62	-0.03–1.07	0.09–0.98	-0.38–1.38
Standard deviation *U*	0.14	0.12	0.10	0.15	0.15	0.15
Coefficient of variation	0.38	0.41	0.50	0.25	0.31	0.45
Mean *V*	-0.034	-0.041	-0.021	-0.110	-0.023	0.085
Mean *W*	-0.012	-0.005	0.002	0.008	-0.012	-0.016
M_3 (s^{-1})	0.053	0.095	0.079	0.078	0.102	0.115
			Lock 5–6			
	Upper	**Mid**	**Lower**	**Upper**	**Mid**	**Lower**
Discharge ($m^3.s^{-1}$)	156.10	206.42	205.74	312.84	367.38	395.22
Transect length (m)	95	213	187	101	225	188
Mean depth (m)	1.97	1.61	2.65	1.96	1.64	2.58
Area (m^2)	397.87	757.44	1187.86	448.16	812.33	1104.12
Mean *U* ($m.s^{-1}$)	0.39	0.27	0.17	0.70	0.45	0.36
U range ($m.s^{-1}$)	-0.08–0.81	-0.05–0.54	-0.27–0.65	0.08–1.16	0.07–0.78	-0.10–0.81
Standard deviation *U*	0.14	0.08	0.10	0.20	0.10	0.11
Coefficient of variation	0.35	0.28	0.63	0.29	0.23	0.29
Mean *V*	-0.056	-0.006	-0.053	0.049	0.033	0.029
Mean *W*	-0.012	-0.005	0.001	-0.011	-0.010	-0.008
M_3	0.068	0.042	0.095	0.067	0.062	0.083

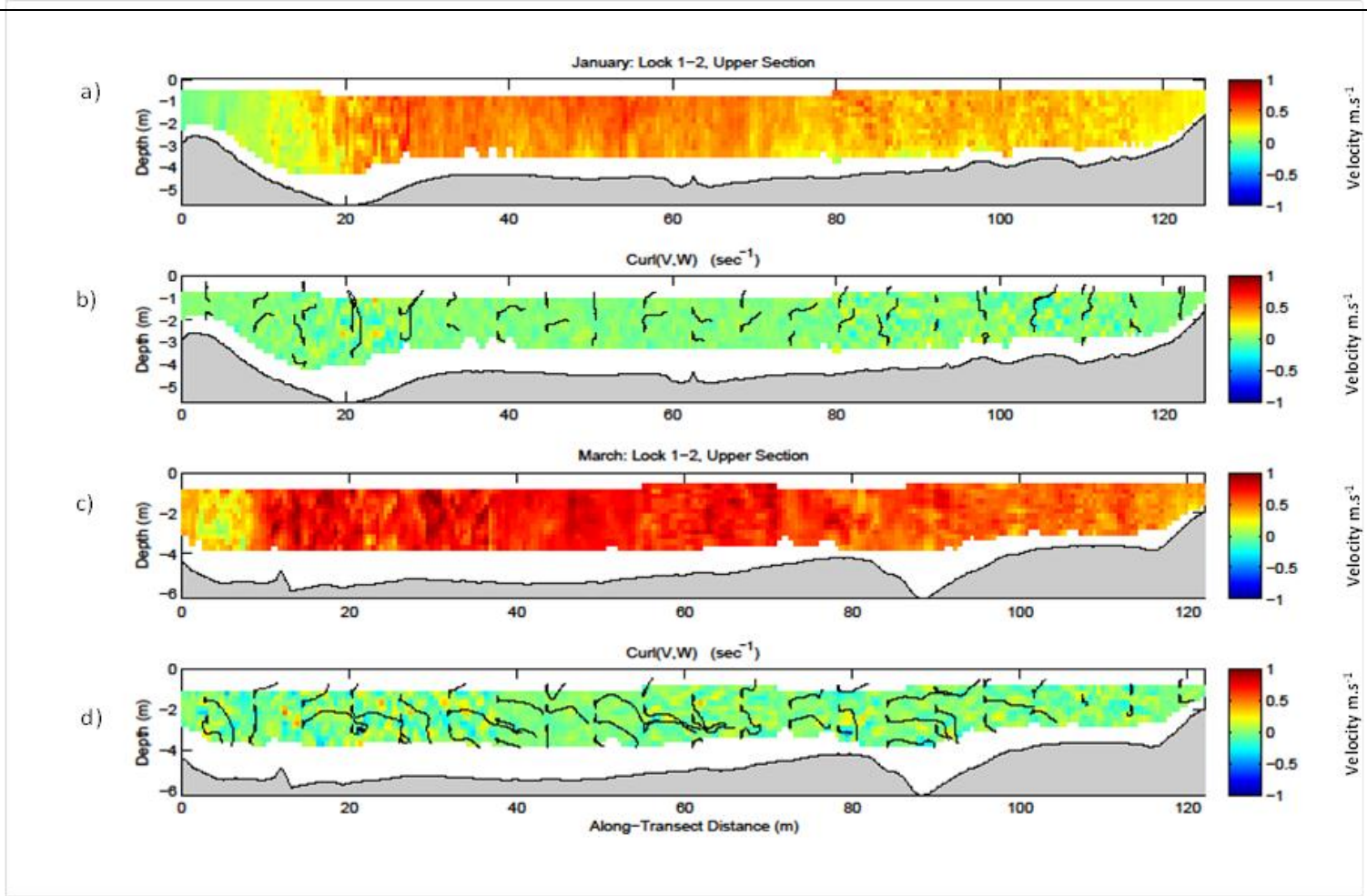


Figure 6. Horizontal current velocity (m.s^{-1}) and circulation (i.e. M_3, curl.s^{-1}) profiles generated for the Lock 1–2 weir pool upper location in January (a & b) and March 2012 (c & d). Current velocity plots (a & c) present cross-transect velocities (U) in cells 0.5 m in width x 0.25 m in height. Circulation plots (b & d) present a combination of velocities in the vertical and horizontal planes (i.e. along-transect, W, and vertically, V), transverse to the banks, with vectors representing the direction of rotation.

Lock 1–2

Mean cross-transect velocities (U) differed significantly between both weir pool location (i.e. upper, middle, lower) (ANOVA; $F_{2, 24864} = 5468.39$, $p < 0.001$) and months (i.e. different flow bands, $F_{1, 24864} = 12{,}171.40$, $p < 0.001$) and there was a significant interaction ($F_{2, 24864} = 5.93$, $p < 0.001$). This indicates that the change in mean velocity between months across weir pool locations were of significantly different magnitudes. Mean velocity was greatest in the upper weir pool and decreased to be least in the lower weir pool location during both flow bands (Figure 7a). Cross-transect velocity ranges increased and mean velocity increased significantly at each weir pool location with increasing flow between January and March (Table 3; Figure 7a). Standard deviation, as a measure of variation in the cross-sectional velocities increased marginally at all locations, however, the coefficient of variation decreased slightly at all sites (Table 3). Additionally, the modified circulation metric (M_3) increased at each location between January and March (Table 3) indicating that the strength and frequency of eddies in river cross-sections was greater under higher flow conditions.

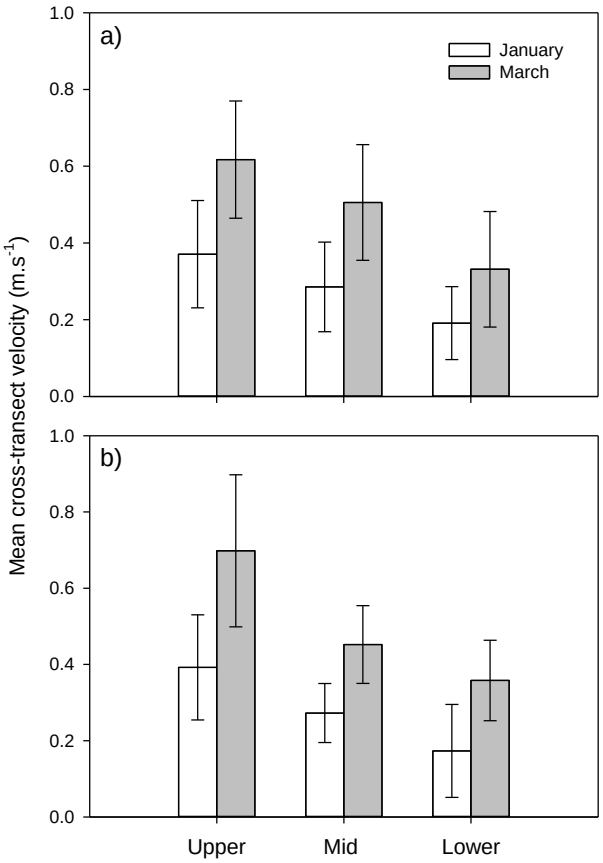


Figure 7. Mean cross-transect velocity (U, m.s^{-1}) ± standard deviation at the upper, mid and lower weir pool location transects of a) the Lock 1–2 reach and b) the Lock 5–6 reach in January and March 2012.

Lock 5–6

Cross-transect velocities differed significantly between both weir pool location (i.e. upper, middle, lower) (ANOVA; $F_{2, 25096} = 8517.79$, $p < 0.001$) and month (i.e. different flow bands, $F_{1, 25096} = 17{,}580.17$, $p < 0.001$) and there was a significant interaction ($F_{2, 25096} = 6.62$, $p < 0.001$), indicating that the change in mean velocity between months across weir pool locations were of different magnitudes. Similar to the Lock 1–2 reach, mean velocity was greatest in the upper weir pool and decreased gradually to be lowest in the lower weir pool during both flow bands (Figure 7b). Cross-transect velocity ranges increased and mean velocity increased significantly at each weir pool location with increasing flow (Table 3; Figure 7b). Standard deviation, as a measure of variation in the cross-sectional velocities increased at all locations

but only marginally in the mid and lower weir pool, whilst the coefficient of variation decreased at all sites (Table 3). The modified circulation metric (M_3) did not increase between January and March in the upper weir pool but increased substantially in the mid weir pool (Table 3). M_3 also appeared to decrease in the lower weir pool location between January and March but this was due to erroneous readings in the January transect at this site which likely elevated the value of M_3.

3.1.3 Comparison of measured vs modelled average velocity

In the Lock 5–6 weir pool, mean velocities derived from DEWNR modelling at QSA of 13,000 and 24,000 ML d^{-1}, and the Water Technology modelling at QSA of 15,000 and 20,000 ML d^{-1}, appeared quite accurate when compared to measured mean velocities at QSA of 16,214 and 33,241 ML d^{-1} (Figure 8a). Water Technology modelling for the upper weir pool at 40,000 ML d^{-1}, however, appeared to underestimate mean velocity, being considerably lower than measured mean velocity at QSA of 33,241 ML d^{-1} (Figure 8a). At QSA of ~13,000 ML d^{-1}, measured and modelled mean velocities were comparable throughout the Lock 1–2 reach (Figure 8b). At the higher discharge of ~24,000 ML d^{-1}, there was greater variation between measured and modelled mean velocities with DEWNR modelling underestimating velocity, particularly in the upper weir pool (Figure 8b).

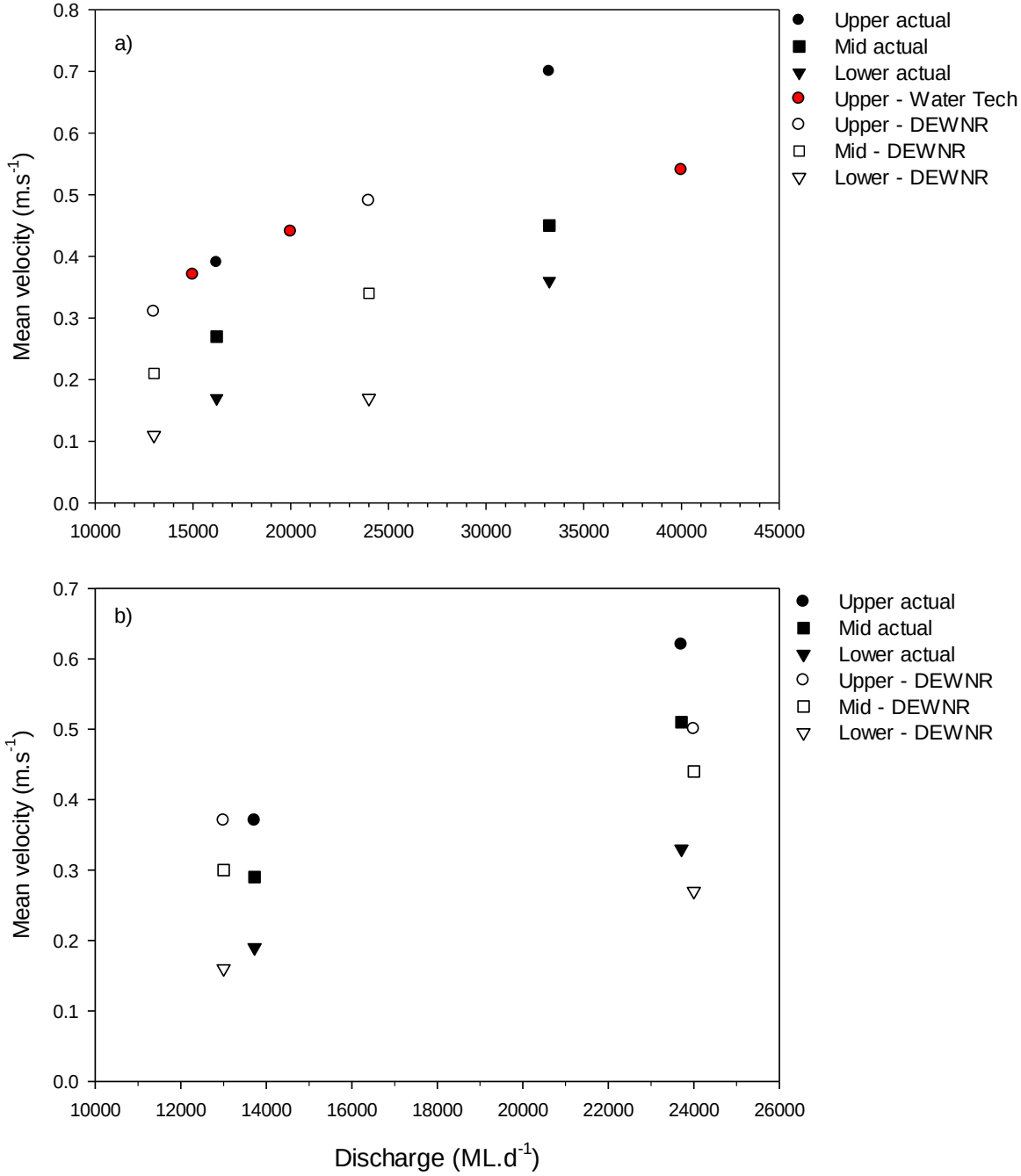


Figure 8. Comparison of mean measured velocities (black symbols) against WaterTech modelled (red symbols; Lock 5–6 upper weir pool only) and DEWNR modelled mean velocities (white symbols) in the upper weir pool (circles), mid weir pool (squares) and lower weir pool (triangles) of a) the Lock 5–6 reach and b) the Lock 1–2 reach under varying discharge (QSA; ML d^{-1}). For the Lock 5–6 reach, QSA is used for simplicity despite the upper weir pool transect being above the Chowilla Creek junction, typically resulting in a lower discharge at this location compared to the mid and lower weir pool, which are downstream of the Chowilla Creek junction.

3.2 Golden perch larvae and larval fish assemblage

3.2.1 Environmental variables

During 2011-12, water discharge mostly remained within-channel (<50,000 ML d^{-1}) in the lower River Murray with flow pulses occurring in early spring and summer (Appendix II). The natural summer flow pulse peaked at ~21,000 ML d^{-1} at both sites (~26,000 ML d^{-1} at the South Australian border) and was supported with the release of a considerable volume of Commonwealth environmental water to the lower River Murray. Discharge in 2011-12 was lower than the previous flood year (i.e. 2010-11) but was much greater than those during 2005-2008. Discharge in 2005-06 exceeded the summer Entitlement Flow (~7,000 ML d^{-1}) but remained within-channel; flow pulses occurred from October to December at both sites with a maximum discharge of ~13,500 ML d^{-1} in November at Site 1 (~15,100 ML d^{-1} at the South Australian border) in the lower River Murray (Appendix II). Due to continuous drought conditions, discharge in 2006, 2007 and 2008 reduced to below Entitlement Flow (Appendix II). In late 2010, a dramatic increase in discharge of natural flows led to overbank flows and extensive flooding in the lower River Murray with flow peaking at >80,000 ML d^{-1} at both sites (~93,000 ML d^{-1} at the South Australian border) in February 2011 (Appendix II).

Water level in 2011-12 largely remained above pool level and the variation pattern was similar to daily discharge at both sites (Appendix III). In previous years, relative water level also generally reflected discharge, particularly at Site 2. At Site 1, relative water level showed a steady decrease from 2006-07 to 2009-10, reaching -1.8 m in July 2009; whilst at Site 2, it remained relatively stable at pool level (Appendix III). In 2010-11, a return to above pool height was recorded for both sites. The increase in discharge in 2010-11 caused a considerable rise in the water level at both sites, resulting in the overbank flood around late November 2011.

Mean daily conductivity varied from 168 to 455 µS cm^{-1} at Site 1 and 141 to 413 µS cm^{-1} at Site 2 during the larval sampling period (October to March) in 2011-12 (Appendix IV). Conductivity was consistently higher at Site 1 than Site 2, with the difference between sites being more pronounced during the low flow yeas (2005-06–2009-10) than high flow/flood years (2010-11 and 2011-12). At Site 1, there was a general trend of increase in conductivity from 2005-06 to 2009-10 followed by a noticeable reduction in 2010-11 and 2011-12 when discharge increased. At Site 2, conductivity mostly varied between 100–300 µS cm^{-1} with a few highs close to 400 µS cm^{-1} in 2007-08, 2008-09 and 2011-12.

Mean daily water temperature ranged between 16 and 27 °C at both sites during the study period from October to March in 2011-12 (Appendix V). Over the last seven years, changes in water temperature reflected seasonal variation. There was little difference between the sites and the seasonal pattern was generally consistent, with temperature increasing steadily from spring to summer.

3.2.2 Catch summary

Ten fish species (nine native and one exotic) were collected in 2011-12 (Table 4). The number of native species was the same as those in 2010-11 and 2005-06, but greater than during the drought years (2006-07 to 2008-09). The number of native species sampled at Site 1 was the highest in 2011-12 compared to the previous five study years.

In 2011-12, the most abundant species collected were the small to medium-bodied native species, Australian smelt, bony herring (*Nematalosa erebi*), carp gudgeon (*Hypseleotris spp*.) and flathead gudgeons (*Philypnodon spp*.), and a large-bodied flow-cued spawning species, golden perch. Less abundant native species collected included hardyhead (*Craterocephalus spp*.), Murray cod (*Maccullochella peelii*), freshwater catfish (*Tandanus tandanus*) and silver perch. For the exotic species, carp (*Cyprinus carpio*) were

sampled in low numbers but no redfin perch (*Perca fluviatilis*) were collected in 2011-12.

Table 4. Summary of species and total number of larval fish sampled at Sites 1 and 2 in the lower River Murray in 2011-12 (shaded) and previous years (2005-06–2010-11 except 2009-10).

Species		Site 1 (D/S Lock 1)						Site 2 (D/S Lock 6)						Total					
Common name	**Scientific name**	**2005-06**	**2006-07**	**2007-08**	**2008-09**	**2010-11**	**2011-12**	**2005-06**	**2006-07**	**2007-08**	**2008-09**	**2010-11**	**2011-12**	**2005-06**	**2006-07**	**2007-08**	**2008-09**	**2010-11**	**2011-12**
Australian smelt	*Retropina semoni*	1,426	6,833	8,951	630	4	407	280	2,763	7,797	3,421	27	66	1,706	9,596	16,748	4,051	31	473
Bony herring	*Nematalosa erebi*	1,045	2,253	117	348	4	865	355	1,002	3,530	357	21	117	1,400	3,255	3,647	705	25	982
Carp gudgeon	*Hypseleotris* spp.	1,487	1,960	4,885	1,991	44	183	589	505	2,486	350	18	102	2,076	2,465	7,371	2,341	62	285
Flathead gudgeon	*Philypnodon grandiceps*	251	3,209	6,674	6,931	74	321	493	1,034	2,336	489	37	159	744	4,243	9,010	7,420	111	480
Hardyhead	*Craterocephalus* spp.	4	-	14	34	-	1	1	20	10	1	1	2	5	20	24	35	1	3
Freshwater catfish	*Tandanus tandanus*	1	-	-	-	1	1	10	4	-	4	1	6	11	4	-	4	2	7
Golden perch	*Macquaria ambigua ambigua*	-	-	-	-	69	530	39	-	-	-	63	105	39	-	-	-	132	635
Murray cod	*Maccullochella peelii*	3	-	-	-	14	1	3	4	2	-	10	1	6	4	2	-	24	2
Silver perch	*Bidyanus bidyanus*	-	-	-	-	5	12	13	-	-	-	1	5	13	-	-	-	6	17
Carp	*Cyprinuns carpio*	3	-	9	109	27	1	15	2	4	3	66	5	18	2	13	112	93	6
Redfin	*Perca fluviatilis*	-	-	5	-	74	-	1	18	-	-	-	-	1	18	5	-	74	-
Eggs						1,955	839					695	81	-	-	-	-	2,650	920
Free embryos						84						9		-	-	-	-	93	-
	Total	**4,220**	**14,255**	**20,655**	**10,043**	**2,355**	**3,161**	**1,799**	**5,352**	**16,165**	**4,625**	**949**	**649**	**6,019**	**19,607**	**36,820**	**14,668**	**3,304**	**3,810**
	Yearly Catch (%)	70.1	72.7	56.1	68.5	71.3	83.0	29.9	27.3	43.9	31.5	28.7	17.0						
	Number of Trips	7	7	6	6	7	11	7	7	6	6	7	11	7	7	6	6	7	11
	No. of fish per trip	**603**	**2,036**	**3,442**	**1,674**	**336**	**287**	**257**	**765**	**2,694**	**771**	**136**	**59**	**860**	**2,801**	**6,137**	**2,445**	**472**	**346**
	No. of species	*8*	*4*	*7*	*6*	*10*	*10*	*11*	*9*	*7*	*7*	*10*	*10*	*11*	*9*	*8*	*7*	*11*	*10*
	No. of native species	***7***	***4***	***5***	***5***	***8***	***9***	***9***	***7***	***6***	***6***	***9***	***9***	***9***	***7***	***6***	***6***	***9***	***9***

3.2.3 Golden perch larval abundance in response to flows

As a flow-cued spawning species, golden perch pre-flexion larvae were identified and used as an indicator of local spawning events or newly spawned drifters from other areas close by. In 2011-12, there was a peak in pre-flexion larvae abundance in October and November at both Site 1 and Site 2, suggesting a major spawning event in spring (Figure 9). This coincided with the falling phase of a natural spring flow of ~35,000 ML d^{-1} and a temperature rise to >18 °C (Figure 9). The discharge of natural flow declined to <10,000 ML d^{-1} in November although some environmental water was released to reduce the rate of recession. Between December 2011 and January 2012, a substantial volume of environmental water was delivered, which enhanced a within-channel natural flow pulse up to 26,000 ML d^{-1} (at the South Australian border) and extended the flow recession during summer (Figure 9). In February, small volumes of additional environmental water delivered to the lower River Murray added to a natural flow increase from ~15,000 to 25,000 ML d^{-1} (at the South Australian border). Additional pre-flexion larvae were sampled in moderate abundance, particularly at Site 1, between late January and early March 2012 (Figure 9). This increased presence of golden perch larvae during summer followed the within-channel flow pulses enhanced by environmental flows (Figure 9).

The size compositions of all golden perch larvae caught each month from Site 1 and Site 2 during the 2011-12 water year are presented in Figure 10, including the late stage larvae (i.e. post-flexion larvae). There were pre- and post-flexion larvae found at both sites with size ranging from 2–14 mm (TL), which probably comprised larvae from locally spawned fish as well as larvae which drifted downstream from the upstream reaches of the Murray–Darling River. The presence of pre-flexion golden perch larvae from October to December at Site 2 and throughout the sampling period at Site 1 suggests spawning activities in the lower River Murray. The numbers of larvae collected were highly

variable between months with November representing 82% of the total catch (Site 1: 69% and Site 2: 13%).

The patterns of golden perch larval abundance in 2011-12 were compared with those in previous years (2005-06 to 2010-11) under different flow scenarios in the lower River Murray (Figure 11). Over six study years, pre-flexion golden perch larvae were only found in the flood and high flow years (2010-11 and 2011-12) (Figure 11); no pre-flexion larvae were found between 2005-06 and 2008-09 when discharge to SA was generally below 15,000 ML d^{-1} (at the South Australian border) (Figure 11). Although some golden perch larvae were collected in 2005-06, all larvae sampled were at later stages of development and therefore not shown on Figure 11. In both 2010-11 and 2011-12, pre-flexion larvae seemed to be more abundant at Site 1, however, no significant difference was detected between sites (p=0.233) or years (p=0.817) for the common sampling period from October to January, probably due to high variability in larval abundance between trips. In both years, there was a peak in larval abundance in late spring/early summer although it occurred slightly earlier in 2011-12 (November) than in 2010-11 (December); pre-flexion larvae continued to be found in January during 2011-12 but not in 2010-11 (Figure 11). During 2011-12, ongoing sampling during the summer following environmental flow releases collected more early stage larvae after January, suggesting potential multiple spawning events in the lower River Murray (Figures 9 and 10).

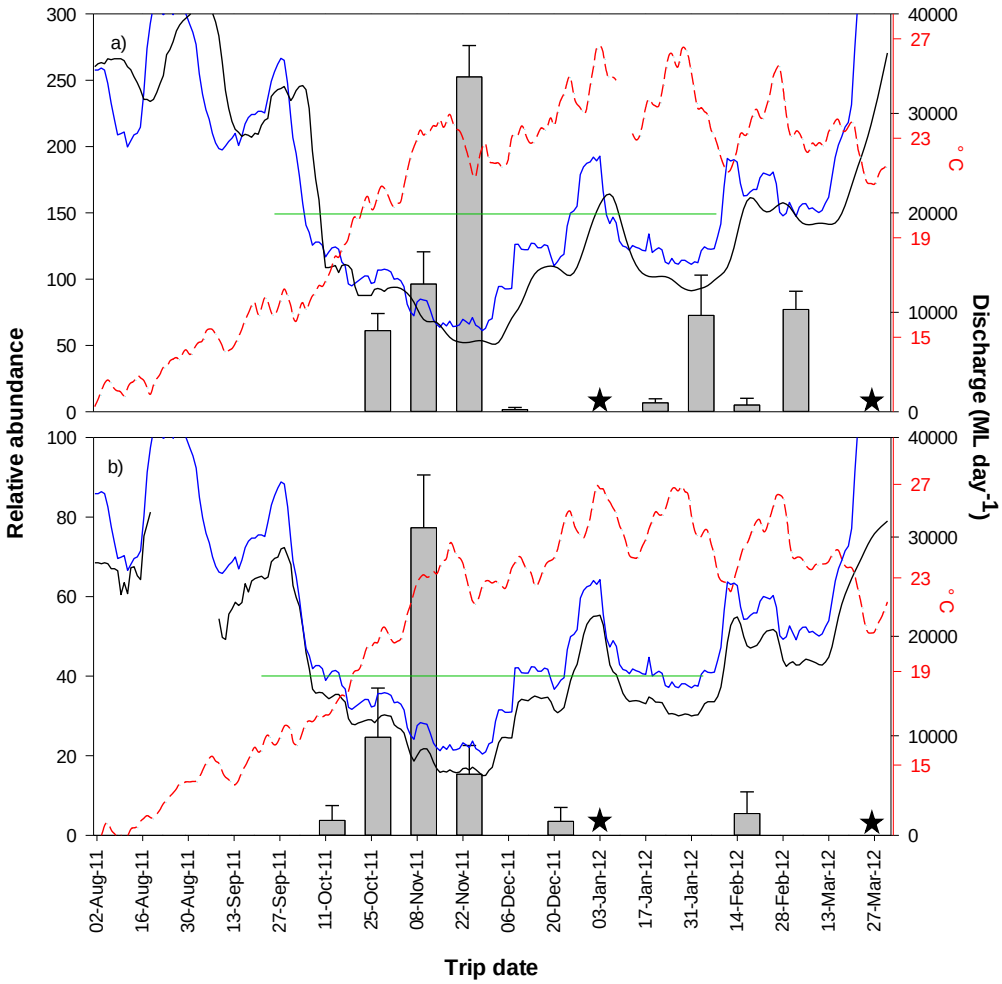


Figure 9. Relative abundance of pre-flexion golden perch larvae sampled in each trip during 2011-12 at Sites 1 (a) and 2 (b) in the lower River Murray. Note scale difference in left Y axis between figures a) and b). Sampling started on October 11 2011. Data presented as the mean of the 6 replicates ± standard error. Water discharge at each site (black line) and at the SA border (blue line) and temperature (red line) are also presented for the sampling period. Green line represents period when Commonwealth environmental water was delivered. Stars indicate that NO sampling occurred during that week.

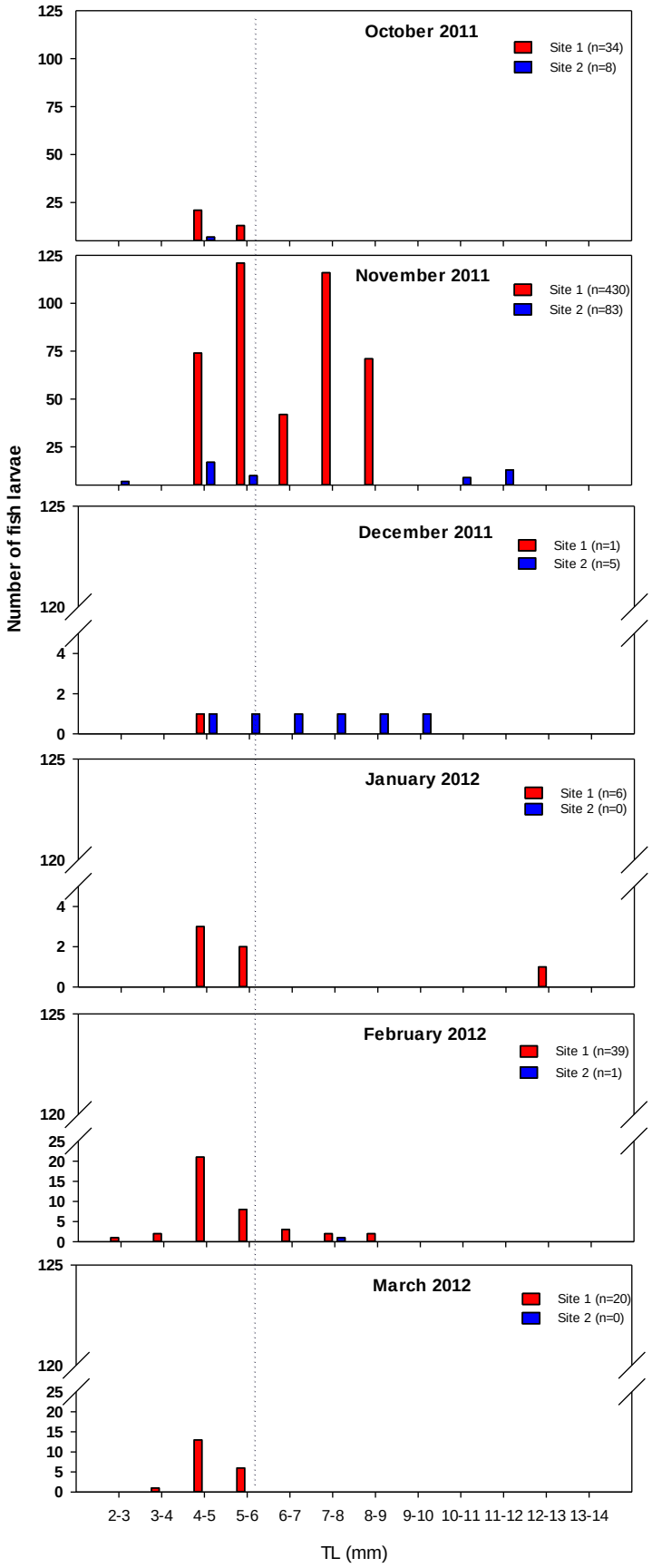


Figure 10. Length frequency distributions of golden perch larvae from October 2011 to March 2012 samples collected at Sites 1 and 2 in the lower River Murray. Dotted line indicates length when flexion of the notochord occurs (i.e. 6.1 mm).

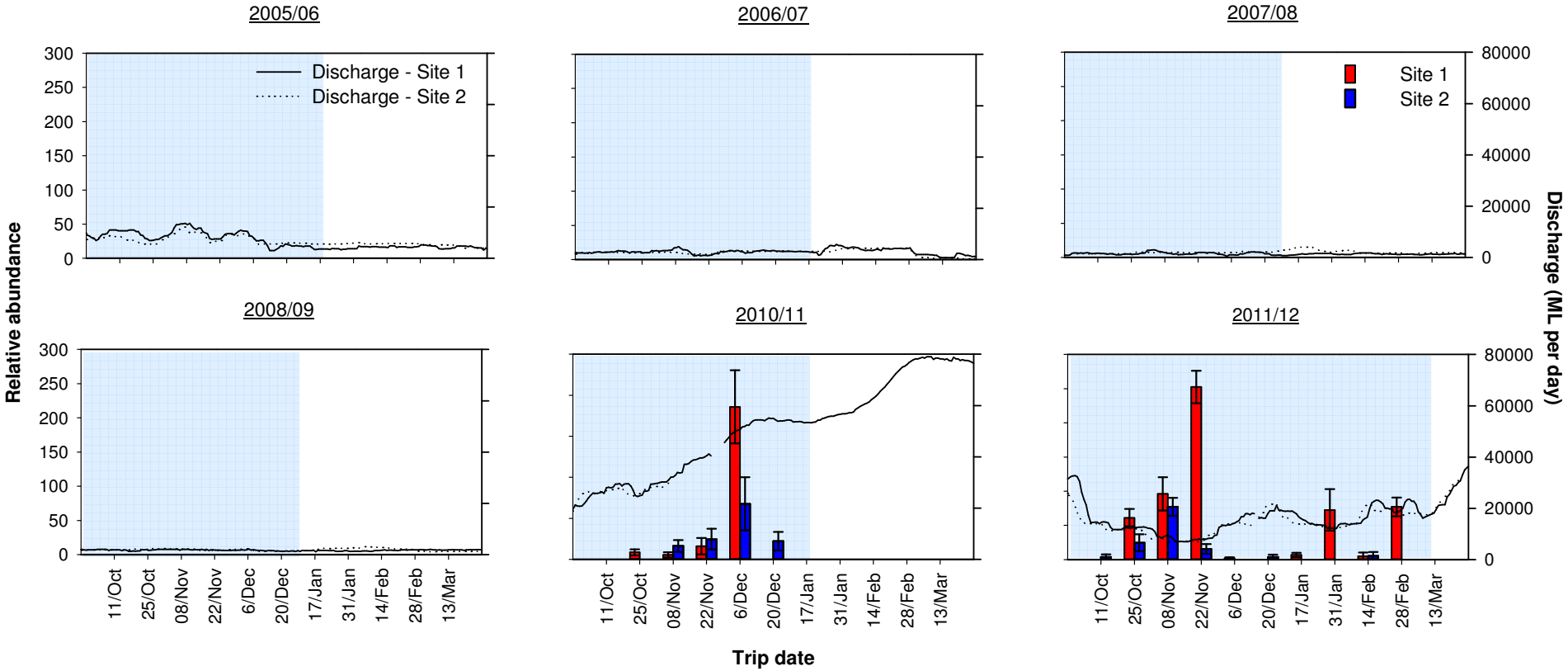


Figure 11. Relative abundance of pre-flexion golden perch larvae sampled in each trip at Sites 1 and 2 in the lower River Murray between 2005-06 and 2011-12 (no sampling in 2009).Data presented as the mean of the 6 replicates ± standard error. Water discharge for both sites is also presented for this period. Temporal extent of sampling season is shaded for each year.

3.2.4 Larval fish assemblages

Principal component analysis indicated that the larval fish assemblage in 2011-12 showed a distinct separation from the drought (2006-07 to 2008-09) years (Figure 12, Appendix VI). Assemblage structure in 2011-12 was more similar to the flood year (2010-11). Interestingly, samples from 2005-06, a low within-channel flow year, interspersed among the samples from drought years (2006-07 to 2008-09) and those from 2011-12. The total variation in larval fish assemblages was well captured by PCO1 and PCO2 (55%)(Figure 12). The four small to medium-bodied species (Australian smelt, carp gudgeon, flathead gudgeon and bony herring) appeared to strongly characterise fish assemblages during the drought years; whereas large-bodied species (golden perch, silver perch, perch free embryos, carp, Murray cod and freshwater catfish) appeared to be more correlated to samples in high within-channel flow and flood years (2010-11 and 2011-12).

In 2011-12, although larval fish assemblages did not differ significantly between Site 1 and Site 2, the pattern of variation compared to previous years differed between the two sites. At Site 1, the difference in the larval assemblage of 2011-12 compared to 2010-11 (the previous flood year) was mainly attributed to the greater abundances of golden perch and the four small to medium-bodied species and the absence of perch hatchlings; whereas at Site 2, it was explained by a substantial reduction in carp, a moderate decline in golden perch and increased abundances of flathead gudgeon, bony herring and carp gudgeon (Appendix VII). Comparing larval fish assemblages between 2011-12 and previous low flow/drought years (2005-06 to 2008-09), the difference was mainly attributed to reduced abundance of the small-bodied species and variation in bony herring abundance at both Site 1 and Site 2 (Appendix VII). However, at Site 2, increased abundance in golden perch in 2011-12 was also an important contributor to the assemblage difference from 2005-06.

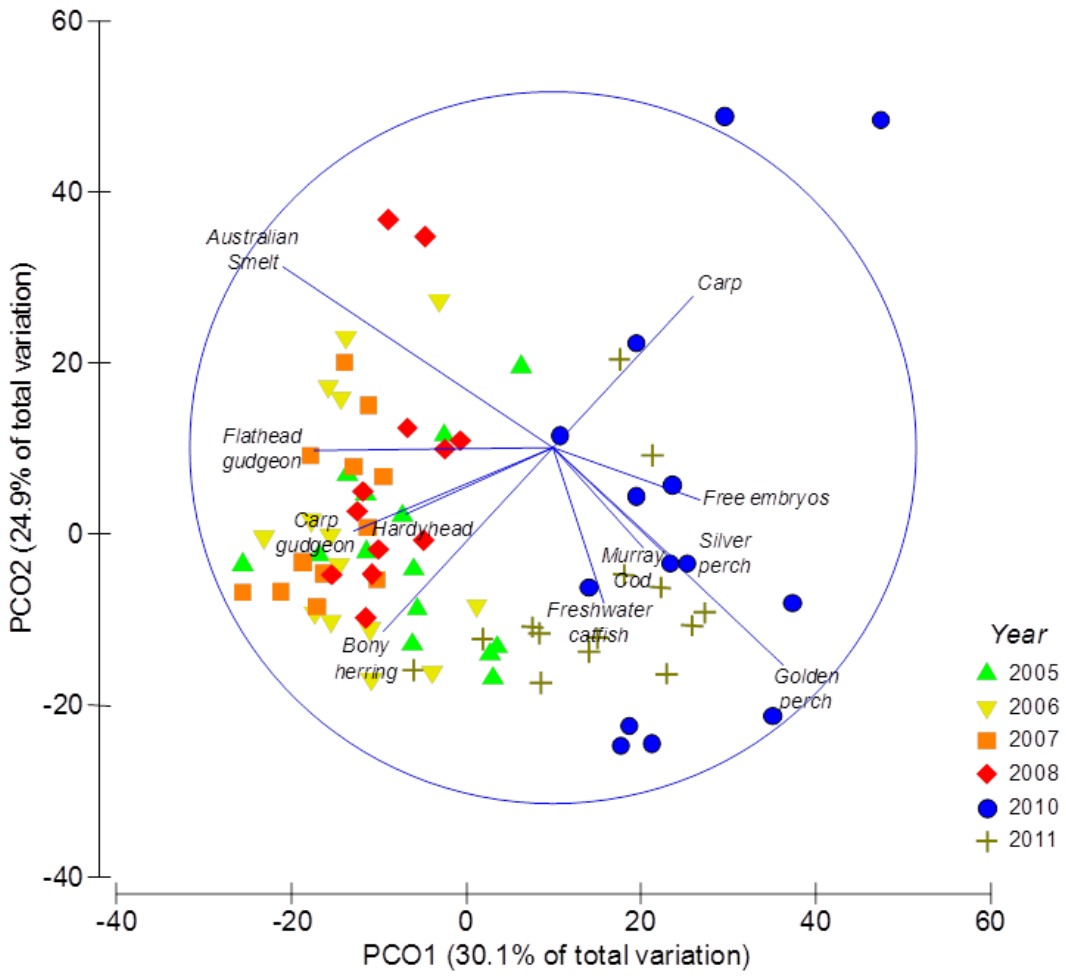


Figure 12. PCO of larval fish assemblage data from 2005 to 2011 in the lower River Murray. It demonstrates the shift in assemblage from small-bodied species dominated in dryer years (2005/06-2008/09) to large-bodied species abundant in the flood year (2010/11) and the following high natural flow year (2011/12) with additional environmental water releases.

3.2.5 Linking environmental variables to larval fish assemblage

Water discharge, temperature and relative water level were the best predictors of larval fish assemblage structure, which together explained 33.8% of the variation throughout the six studied years; although conductivity was also identified as a significant factor; adding it only improved the proportion of the variation explained to 36.6% (Appendix VIII). The horizontal distribution of the samples from the drought (2006-07 to 2008-09) to the within-channel flow years (2005-06, 2011-12) and the flood year (2010-11) was well explained by the positive correlation with water discharge and relative water level. The vertical distribution of

samples from all years was primarily driven by temperature and reflected the seasonal variation of the samples (Figure 13).

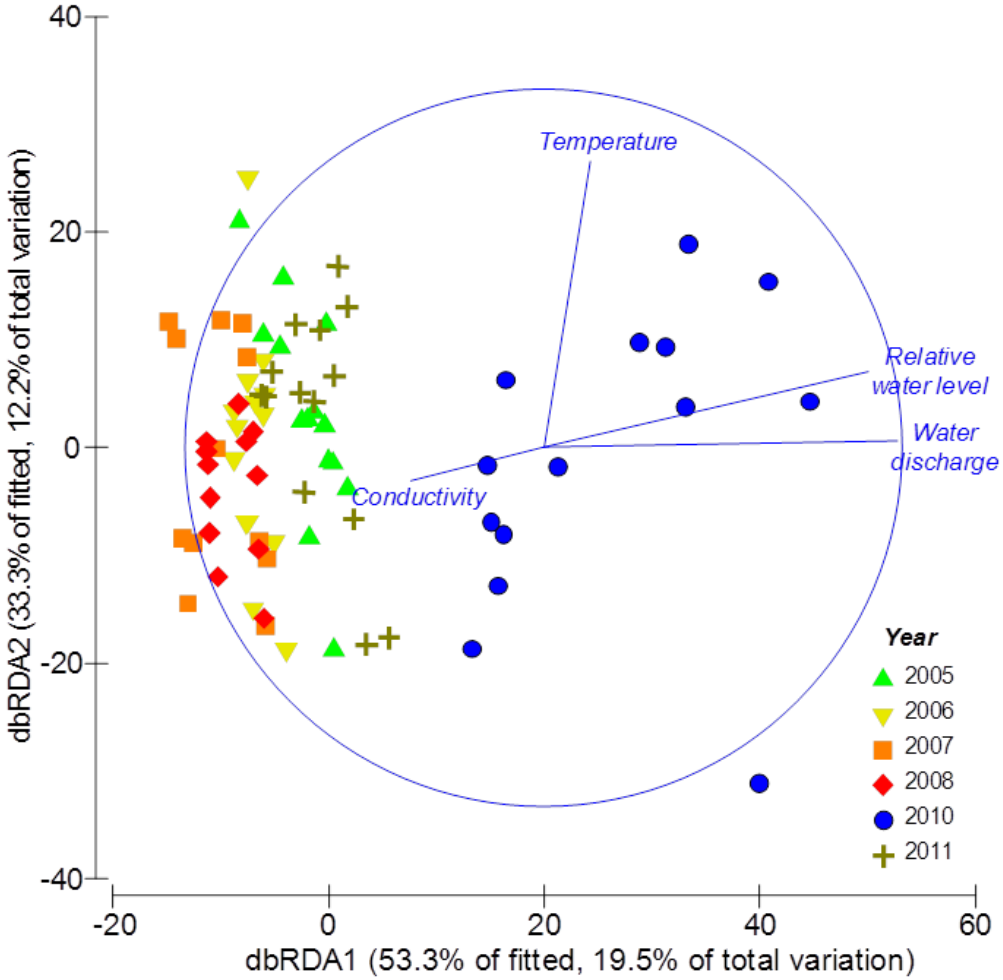


Figure 13. dbRDA ordination of the fitted model of larval fish assemblage data versus the environmental predictor variables, water discharge, relative water level, temperature and conductivity in the lower River Murray. It demonstrates that the horizontal separation of larval assemblages in 2011/12 and 2010/11 from those of the drought years (2006/07-2008/09) was well explained by the increases in discharge and relative water level.

3.3 Recruitment and natal origin of golden perch

3.3.1 Hydrology

From July to October 2011, flow in the lower River Murray (discharge at the South Australian border, QSA) generally reflected flow in the mid-reaches of the Murray at Euston. In both reaches, an extended (~3 months) within-channel bank-full flow event peaked at approximately 40,000 ML d^{-1} in August/September 2011 before receding to approximately 10,000 ML d^{-1} in November 2011. Flow in the Darling River was <1,000 ML d^{-1} throughout this period (Figure 14). Between December 2011 and March 2012, flow in the River Murray at Euston decreased to approximately 5,000 ML d^{-1} while flow in the Darling at Burtundy increased to 15,000 ML d^{-1} and flow in the lower Murray ranged 15–25,000 ML d^{-1} as a result of flow from the Darling and water released from Lake Victoria (Figure 14).

From mid November 2011 to February 2012 the Commonwealth Environmental Water Holder and the MDBA traded environmental water from Goulburn, Murrumbidgee and Campaspe catchments to the lower River Murray. From November 2011 to January 2012, 131 GL of Commonwealth environmental water was delivered to the River Murray in South Australia at a rate of ~1,000–5,000 ML d^{-1}, this water was traded from the Goulburn and Campaspe rivers but sourced from Lake Victoria (Figure 14). From January to February 2012, 152 GL of Commonwealth environmental water was delivered primarily from the Darling River at a rate of approximately 4,000–5,000 ML d^{-1} (CEWO, pers comm).

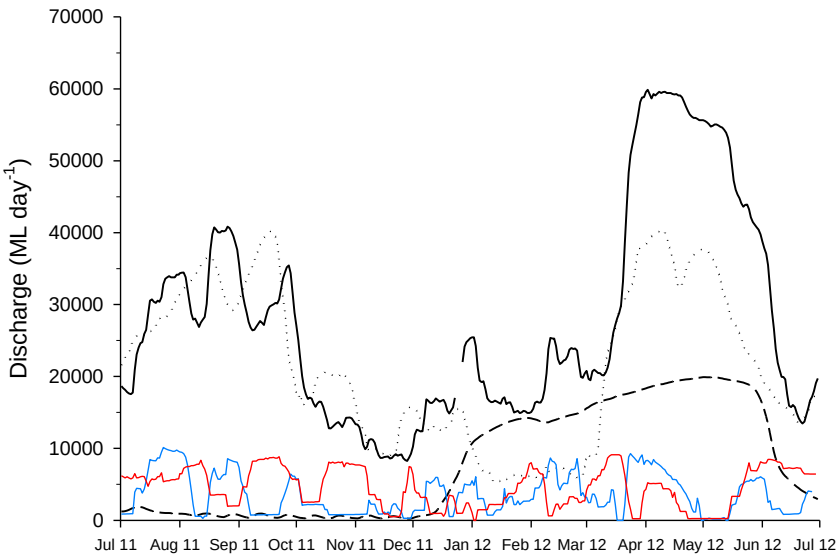


Figure 14. Discharge (ML d^{-1}) in the River Murray at the South Australian border (solid black line) and Euston (dotted black line), Darling River at Burtundy (dashed black line), Frenchman's Creek (Lake Victoria inlet, solid red line) and Rufus River (Lake Victoria outlet, solid blue line).

3.3.2 Golden perch length and age structure

In 2012, golden perch populations sampled in the floodplain, gorge, swamplands and lakes geomorphic regions of the lower River Murray were dominated by 1 and 2 year old fish spawned in 2010-11 and 2009-10 respectively (Figure 15). Proportions of these age-classes were highest in the floodplain region where age 1+ and 2+ fish contributed to 89% of the sampled population.

In the floodplain and gorge regions, YOY (age 0+) fish, spawned in 2011-12, comprised 9% and 6% respectively, of the sampled populations and this age class was absent from the lower lakes and swamplands (although only 7 golden perch in total were collected from this region). The remainder of the sampled populations were comprised of 6, 11 and 15 year old fish fish, spawned in 2005, 2000 and 1996 respectively and these cohorts generally represented <15% of the sampled populations (Figure 15).

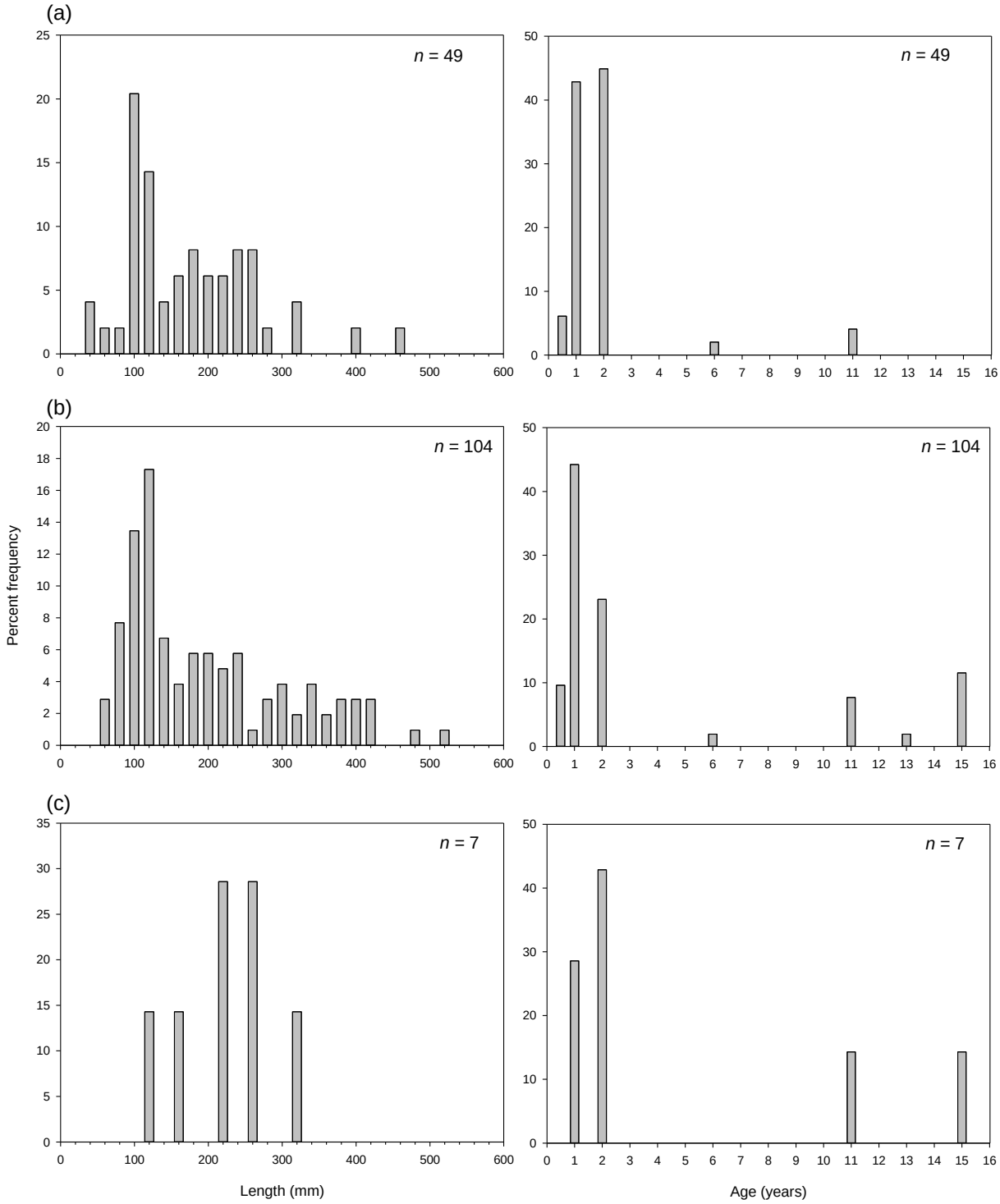


Figure 15. Length (left column) and age (right column) frequency distribution of golden perch collected by boat electrofishing from the (a) floodplain, (b) gorge and (c) swamplands and lakes geomorphic regions of the River Murray in March/April 2012.

3.3.3 Water and otolith $^{87}Sr/^{86}Sr$ signatures and spawn dates of YOY golden perch

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

When water sample collection began in mid-January 2012 a substantial proportion of flow in the lower River Murray at the South Australian border (QSA) was derived from the Darling River and $^{87}Sr/^{86}Sr$ at Lock 9 and Lock 6 decreased towards the Darling River value (nominally a temporally stable 0.7075), with a time lag at Lock 1 (Figure 16). As flows in the mid Murray significantly increased in March 2012, $^{87}Sr/^{86}Sr$ values again increased, exhibiting a mixed Murray and Darling signature (Figure 16).

Spawn dates and otolith $^{87}Sr/^{86}Sr$

Back-calculated spawn dates of fifteen YOY golden perch indicated spawning between mid-October 2011 and early January 2012 (Figure 16). Four fish, spawned between 16 October and 11 November 2011, had otolith core $^{87}Sr/^{86}Sr$ values indicative of the River Murray (Figure 16) but in the absence of water samples from this period we cannot establish if the spawning location was upstream or downstream of the Darling River junction. A further 11 fish were spawned between the 27 November 2011 and 1 January 2012, with ten fish having otolith core $^{87}Sr/^{86}Sr$ values indicative of the Darling River and one fish exhibiting a distinctly different $^{87}Sr/^{86}Sr$ value (0.7116), most likely a mixed Murray and Darling signature, indicative of being spawned in the River Murray downstream of the Darling junction (Figure 16).

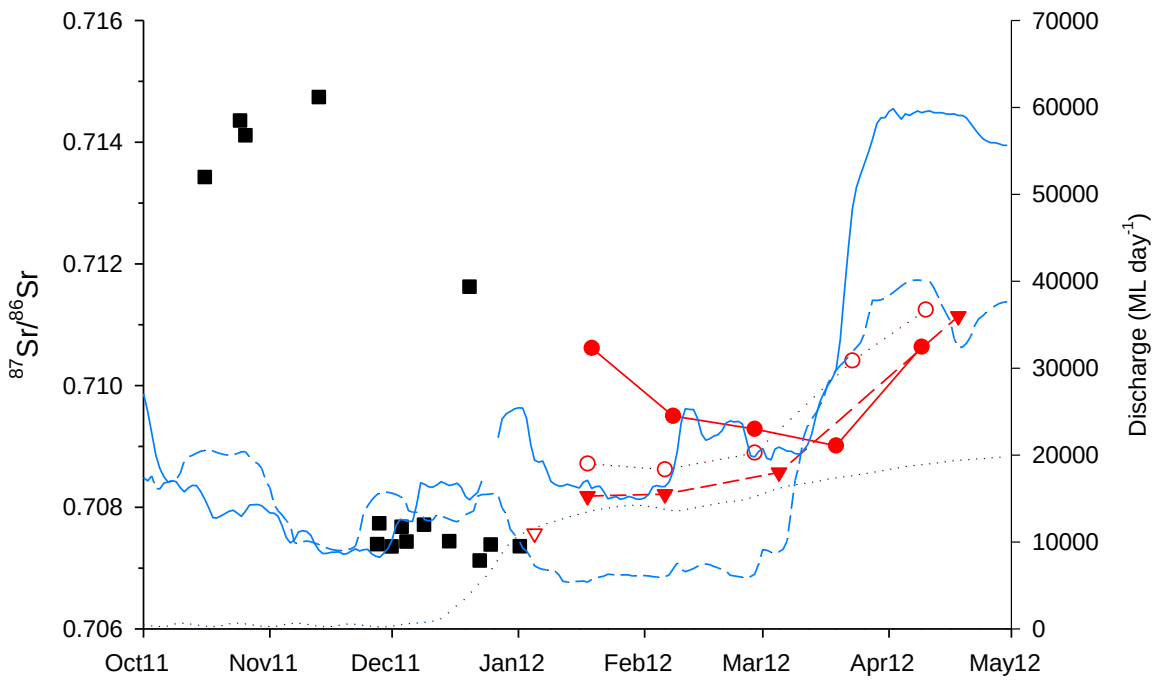


Figure 16. Mean daily discharge in the River Murray at the South Australian border (solid blue line) and Euston (dashed blue line), and Darling River at Burtundy (dotted blue line). $^{87}Sr/^{86}Sr$ ratios in water samples collected from mid-January to mid-April 2012 in the River Murray at Lock 9 (solid red triangles), Lock 6 (open red circles) and Lock 1 (solid red circles), and early-January in the Darling River (open red triangle). Closed black squares represent spawn date and otolith core $^{87}Sr/^{86}Sr$ ratio of golden perch (*n* = 15) collected in the lower River Murray from December 2011 to February 2012.

Life-history profiles for three YOY golden perch, based on transects of $^{87}Sr/^{86}Sr$ from the otolith core to edge, show that two fish spawned in the Darling River (GP 243 and GP 231) exhibit modulation of $^{87}Sr/^{86}Sr$ only mid-late in the fish's lives thus potentially indicating that these fish spent a substantial proportion of their early life in the Darling River before entering the River Murray (Figure 17). These fish were 52 (GP 243) and 66 (GP 231) days old when collected in the gorge and floodplain geomorphic regions of the lower Murray respectively. A third fish (GP 227) exhibits a life-history $^{87}Sr/^{86}Sr$ profile characteristic of being spawned in the River Murray and remaining in this environment as it moves (passively and/or actively) downstream with $^{87}Sr/^{86}Sr$ reflecting the spatio-temporal variability that would be expected in the lower

River Murray (Figure 17). This fish was 67 days old when collected in the gorge geomorphic region of the lower Murray.

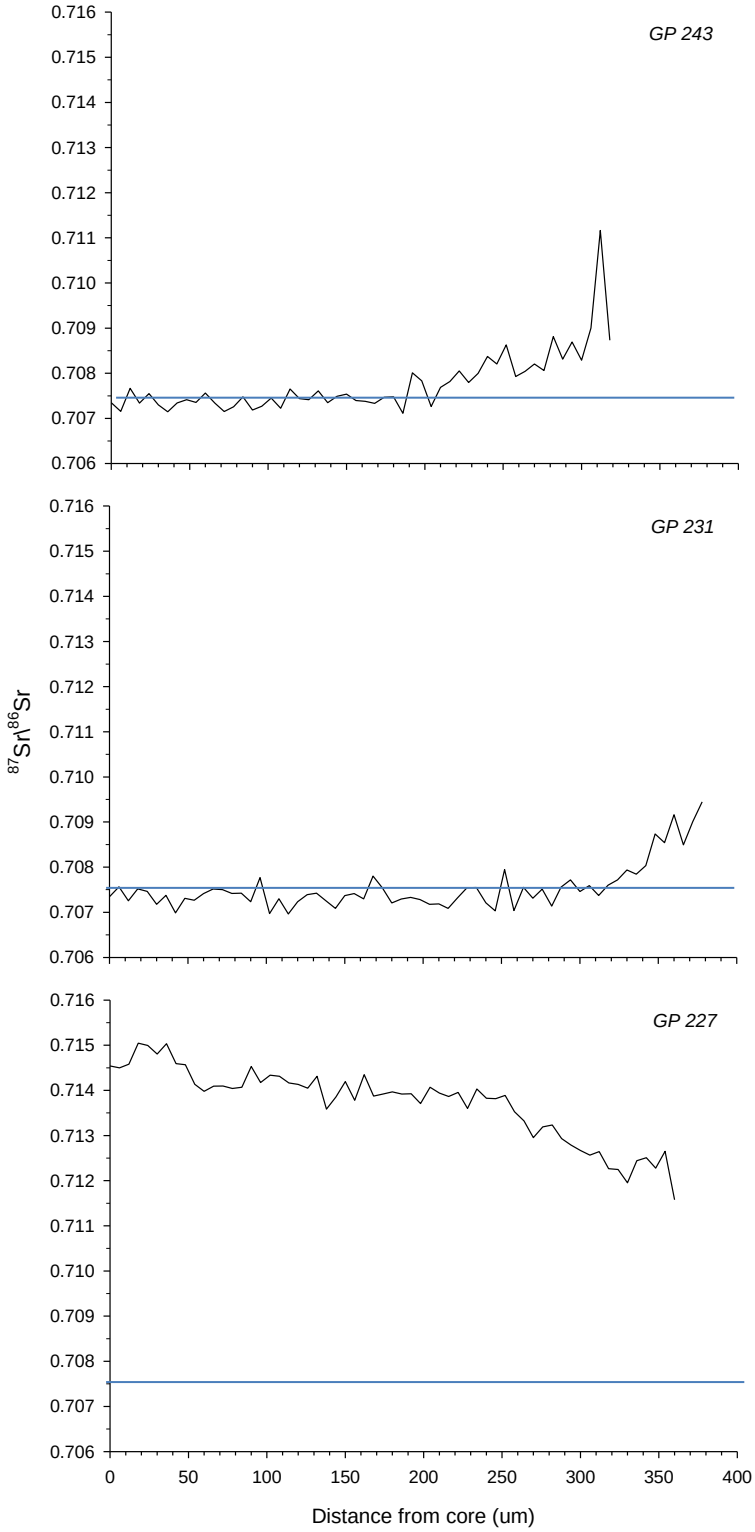


Figure 17. Individual life history profiles based on otolith Sr isotope transects (core to edge) for three juvenile golden perch aged 52 (GP 243), 66 (GP 231) and 67 (GP 227) days. Solid blue lines denote $^{87}Sr/^{86}Sr$ ratio for Darling River water.

Otolith core $^{87}Sr/^{86}Sr$ was also measured in 34 YOY fish that were collected from the floodplain (*n* = 13), gorge (*n* = 16), and swamplands/lower lakes (*n* = 5) geomorphic regions of the lower Murray in 2012 for which no spawn date was determined (Table 5). In total, 22 (65%) of the fish had core $^{87}Sr/^{86}Sr$ values indicative of a Darling River origin and 12 (35%) a River Murray origin. The ratios of Darling to Murray spawned fish were similar in the floodplain and gorge geomorphic regions, 9:4 and 12:4, respectively, but River Murray spawned fish dominated (1:4) the small sample of YOY fish from the swamplands/Lower Lakes region.

Table 5. Otolith core $^{87}Sr/^{86}Sr$ measured in 34 young-of-year golden perch collected from the floodplain, gorge and swamplands/lower lakes geomorphic regions of the lower River Murray.

Geomorphic region	Capture date	$^{87}Sr/^{86}Sr$
Floodplain	5/03/2012	0.70716
Floodplain	6/03/2012	0.70725
Floodplain	6/03/2012	0.70738
Floodplain	6/03/2012	0.71403
Floodplain	14/03/2012	0.70756
Floodplain	2/04/2012	0.71349
Floodplain	2/04/2012	0.71389
Floodplain	6/11/2012	0.70730
Floodplain	6/11/2012	0.70748
Floodplain	6/11/2012	0.70751
Floodplain	6/11/2012	0.71200
Floodplain	7/11/2012	0.70732
Floodplain	7/11/2012	0.70749
Gorge	11/04/2012	0.71241
Gorge	17/05/2012	0.71259
Gorge	22/10/2012	0.70724
Gorge	22/10/2012	0.70725
Gorge	24/10/2012	0.70727

Gorge	24/10/2012	0.70728
Gorge	24/10/2012	0.70732
Gorge	24/10/2012	0.70743
Gorge	24/10/2012	0.70746
Gorge	24/10/2012	0.70761
Gorge	19/11/2012	0.70714
Gorge	19/11/2012	0.71203
Gorge	19/11/2012	0.71222
Gorge	20/11/2012	0.70721
Gorge	20/11/2012	0.70730
Gorge	21/11/2012	0.70747
Swampland/Lower Lakes	7/11/2012	0.71166
Swampland/Lower Lakes	8/11/2012	0.70753
Swampland/Lower Lakes	8/11/2012	0.71173
Swampland/Lower Lakes	8/11/2012	0.71220
Swampland/Lower Lakes	9/11/2012	0.71232

3.4 Salt and nutrient transport

3.4.1 Impact of environmental flows on salinity concentrations

Owing to the existing low salinity levels and moderate flows in the lower River Murray during the study period, the modelling suggests that environmental water provisions had little effect on salinity levels in the lower River Murray (Figure 18). However, further downstream (Lower Lakes and Coorong) salinity was reduced. This was particularly evident at the Murray Mouth and Northern Coorong, with modelling suggesting that electrical conductivity was up to approximately 18,000 µS/cm lower with environmental water provisions during February 2012 (equivalent to reduction in electrical conductivity of up to approximately 70%). This appeared to be a result of increased flows (of lower salinity water) through the Murray Mouth and a reduction in seawater incursions. These findings suggest that environmental water

can be used to reduce seawater incursions and in doing so reduce salinity concentrations in the Northern Coorong. Although environmental water provisions only decreased salinity levels slightly elsewhere, this was most likely due to the antecedent high flow conditions that had already flushed salt from the system. If environmental water was provided during periods of low river flow, when salt tends to accumulate in the system (see Mosley *et al.* 2012 and Aldridge *et al.* 2012), it is likely that the impacts of environmental water provisions on salinity levels would be much greater in the Lower Lakes and lower River Murray than were observed in this study.

3.4.2 Impact of environmental flows on salt transport

Based on modelling outputs, environmental water provisions increased salt exports from the River Murray, Lower Lakes and Murray Mouth, with the greatest effect occurring in February 2012 (Figure 19). The modelling suggests that in February 2012, environmental water accounted for approximately 25%, 25% and 70% of the total salt exports from the River Murray, Lower Lakes and Murray Mouth, respectively. These findings suggest that if sufficient environmental flows are provided, the impact on salt export from the system can be significant. Salt export is a function of the salt concentrations and discharge and because the influence on salinity concentration was only small, much of the difference in cumulative loads associated with environmental flow provisions was attributed to increased discharge through the system.

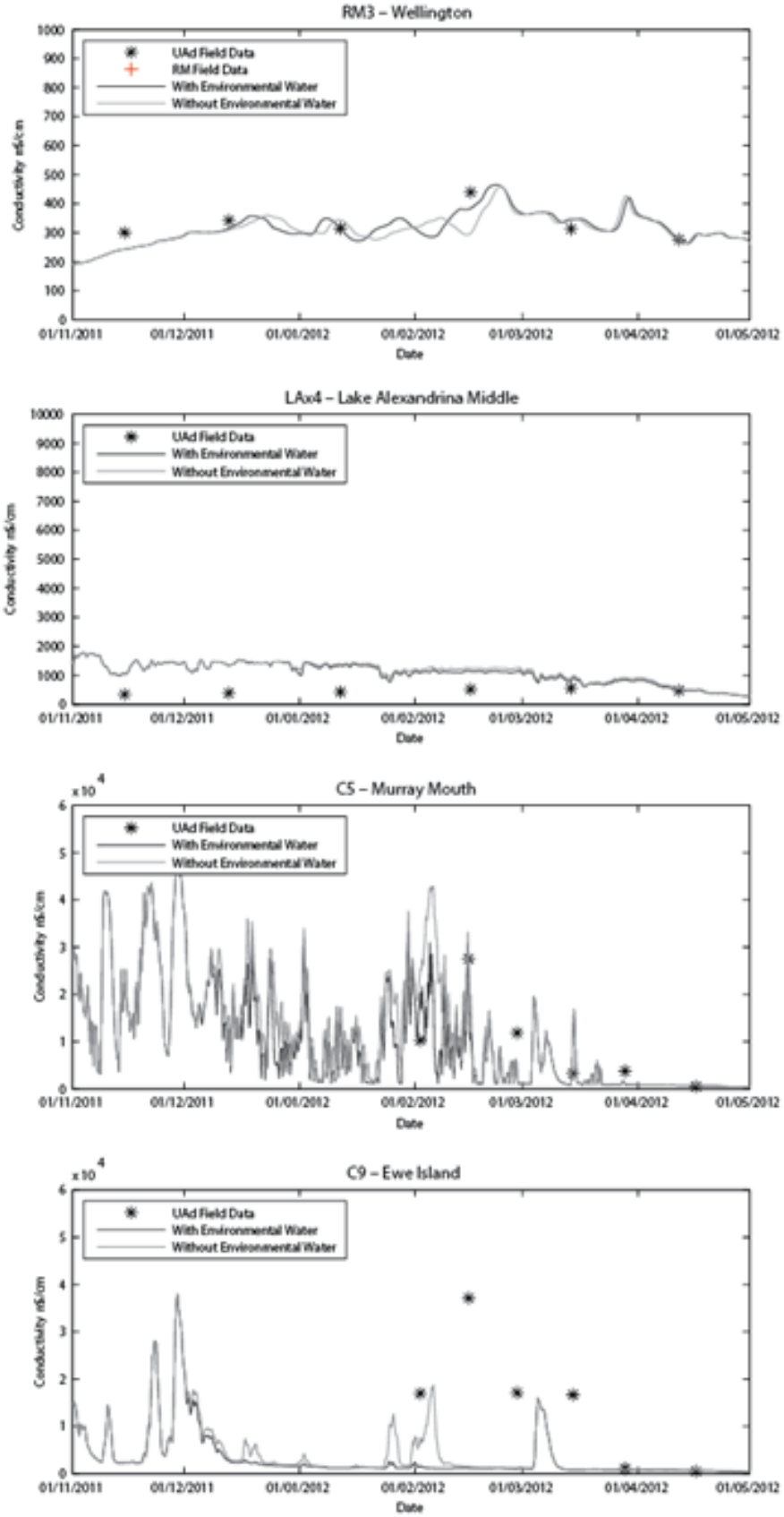


Figure 18. Observed and modelled (with and without environmental water provisions) electrical conductivity data for selected sites.

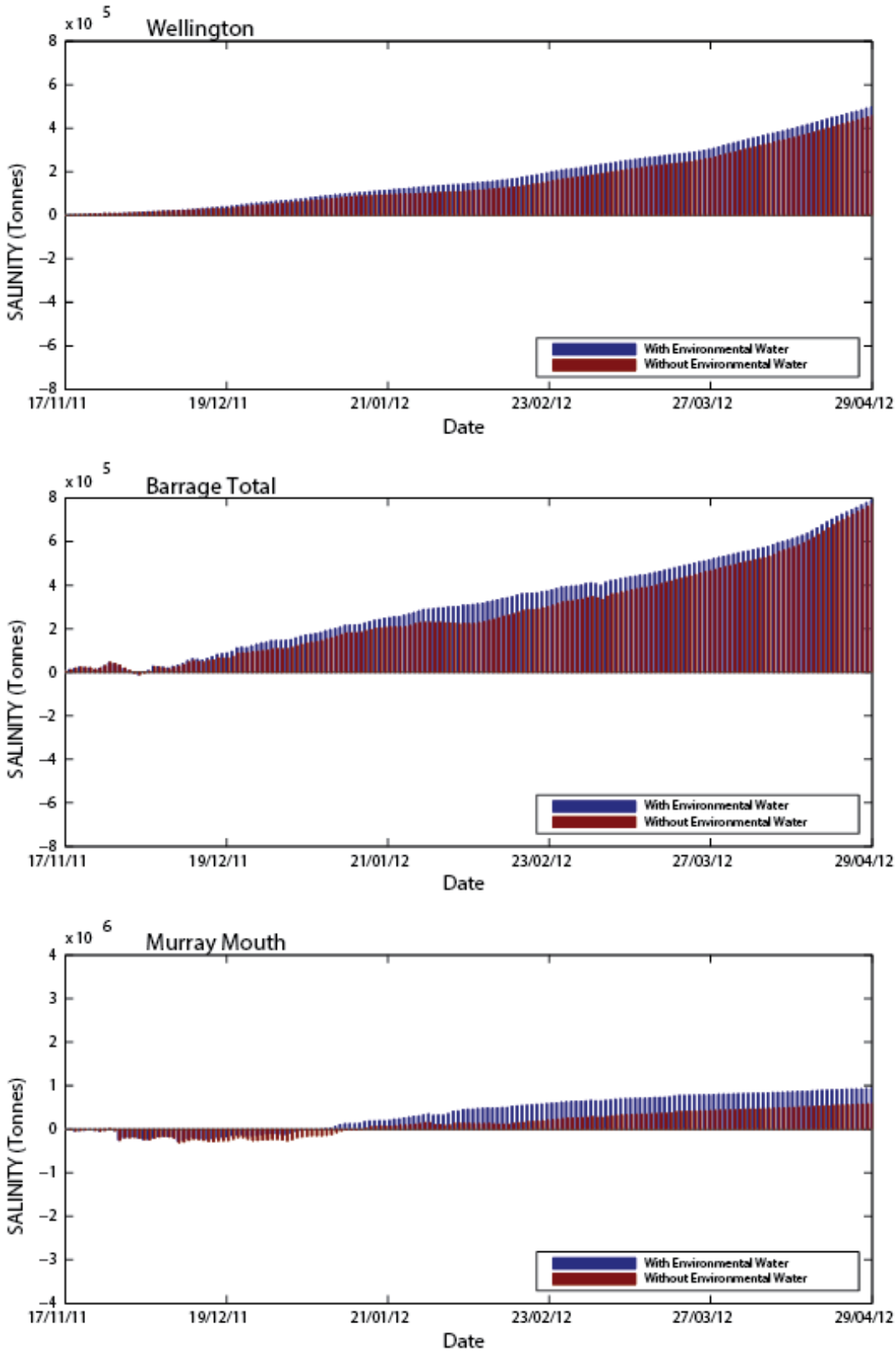


Figure 19. Modelled salt exports with and without environmental flow provisions. Positive values are exports from the River Murray to Lake Alexandrina (Wellington); from the Lower Lakes to the Coorong (Barrage Total); and from the Murray Mouth to the Southern Ocean (Murray Mouth).

3.4.3 Impact of environmental flows on nutrient concentrations

There were only small differences in the modelled dissolved nutrient concentrations with and without environmental water (Figure 20). This was also the case for particulate organic nutrient concentrations upstream of the barrages (Figure 21). However, in the Coorong, particulate organic nutrient concentrations were predicted to be higher with environmental water than without. During the period of environmental water, in the Murray Mouth organic nitrogen concentrations were approximately 0.5 mg/L greater with environmental water provisions than without (concentrations without were between 1 and 2 mg/L). Similarly, organic phosphorus concentrations were approximately 0.05 mg/L greater with environmental water provisions than without (concentrations without were between 0.8 and 1.5 mg/L). This was most likely associated with reduced inputs of seawater (low concentrations) and increased inputs of water from the Lower Lakes (high concentrations). In the Lower Lakes, dissolved nutrients are readily incorporated into the biomass of organisms, resulting in the transport of organic nutrients to the Coorong (Cook *et al.* 2010). The findings of this study suggest that environmental water provisions can be used to export water with elevated organic concentrations from the system to the Southern Ocean, which may increase the productivity of the near-shore environment.

3.4.4 Impact of environmental flows on nutrient transport

Based on modelling outputs, environmental water provisions were predicted to increase exports of dissolved nutrients from the River Murray to the Lower Lakes, but further downstream there was little influence (Figure 22). This was due to the apparent transformation of dissolved nutrients (including incorporation into organic material) within the Lower Lakes and Coorong. This transformation meant that environmental water provisions resulted in increased export of particulate organic nutrients into the Southern Ocean (Figure 23). This conversion will have important implications for the productivity of

downstream ecosystems, as described by Cook *et al.* (2010), but this was not assessed in this study. As observed with salinity, the greatest effect of the environmental water provisions on nutrient transport occurred in February 2012. The modelling suggests that in February 2012, environmental water accounted for approximately 30%, 25% and 50% of the particulate organic nutrient exports from the River Murray, Lower Lakes and Murray Mouth, respectively.

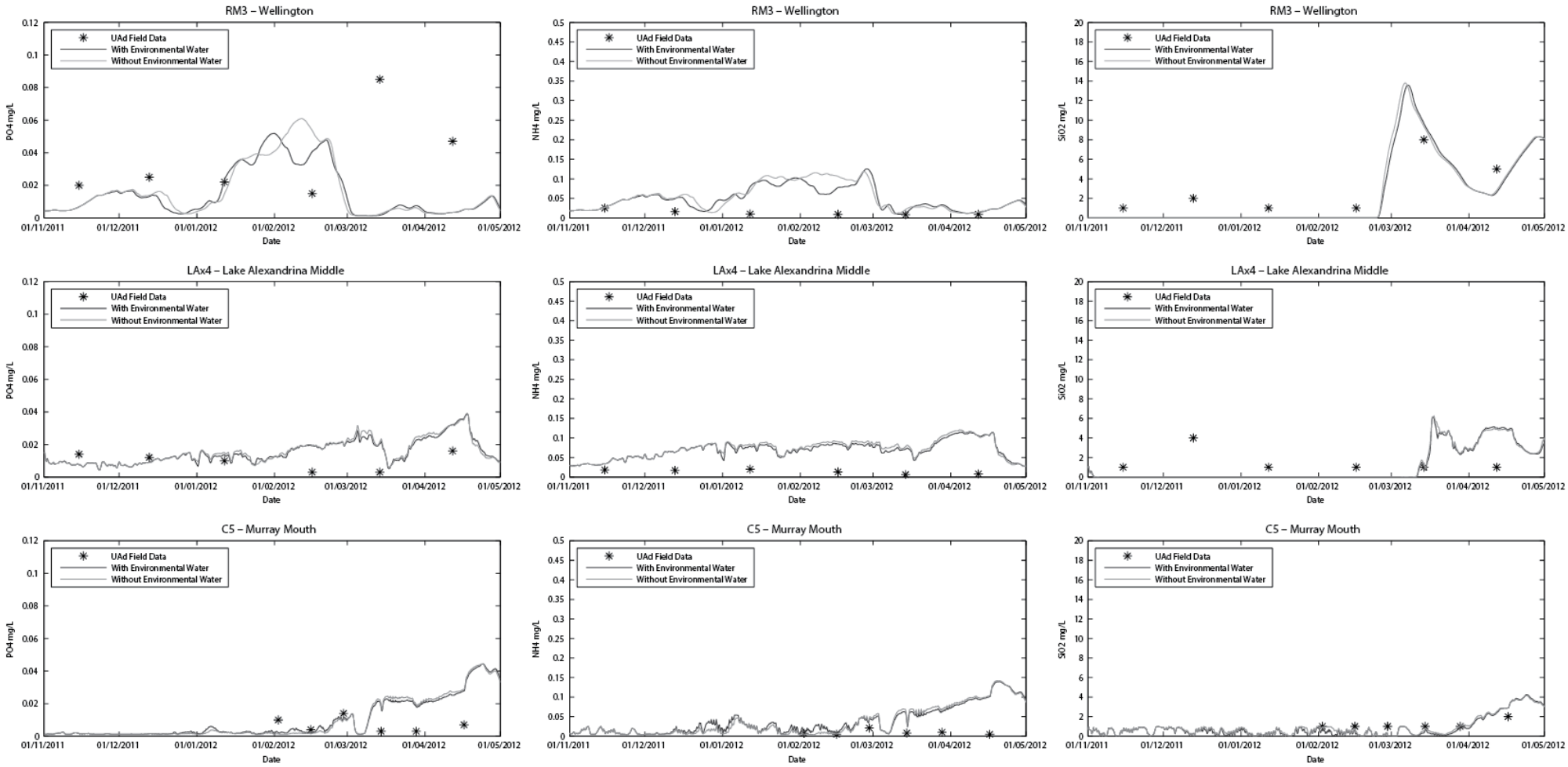


Figure 20. Observed and modelled (with and without environmental water provisions) phosphate (PO_4) ammonium (NH_4) and silica (SiO_2) concentrations at selected sites.

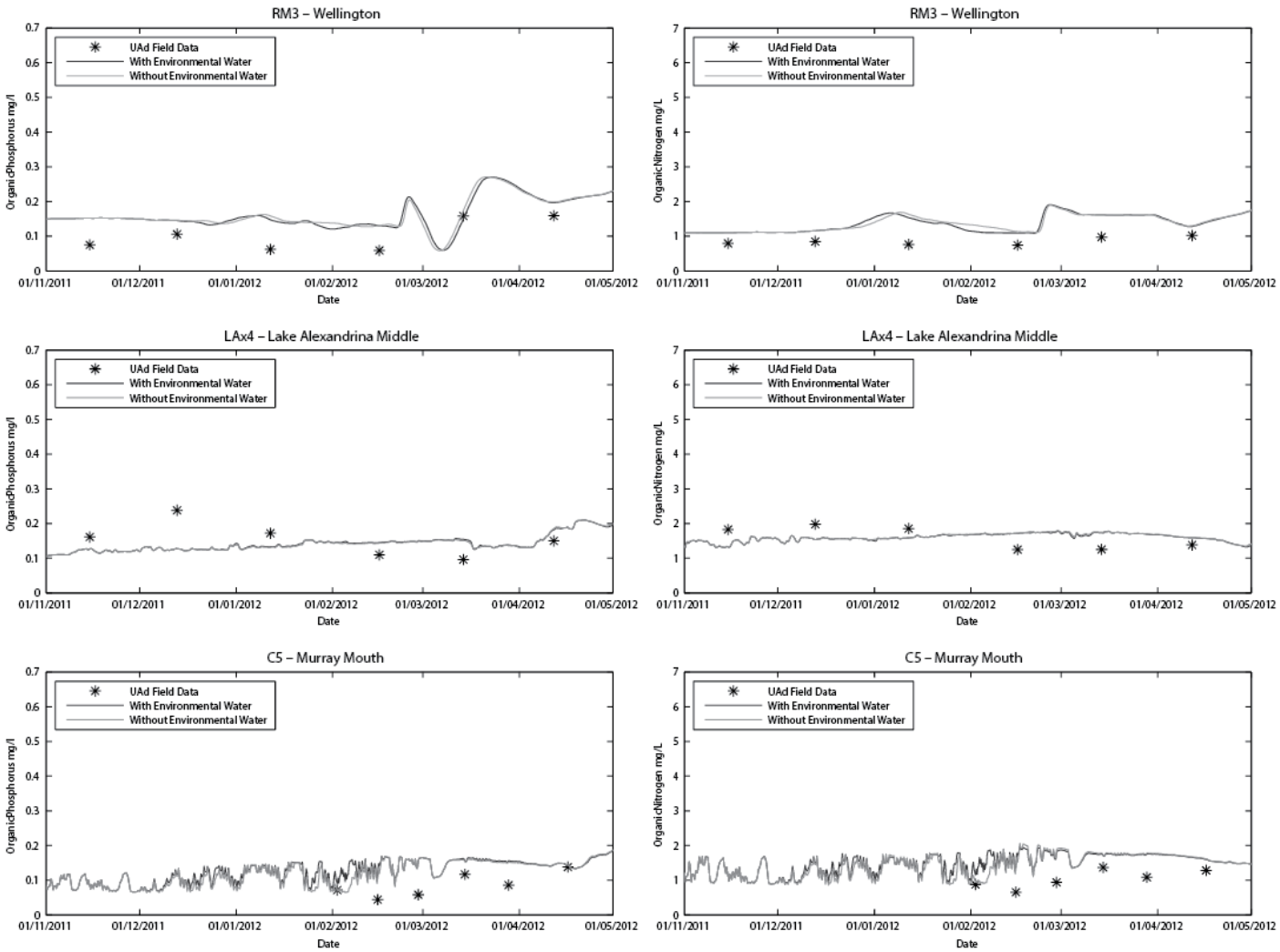


Figure 21. Observed and modelled (with and without environmental water provisions) particulate organic phosphorus and nitrogen concentrations at selected sites.

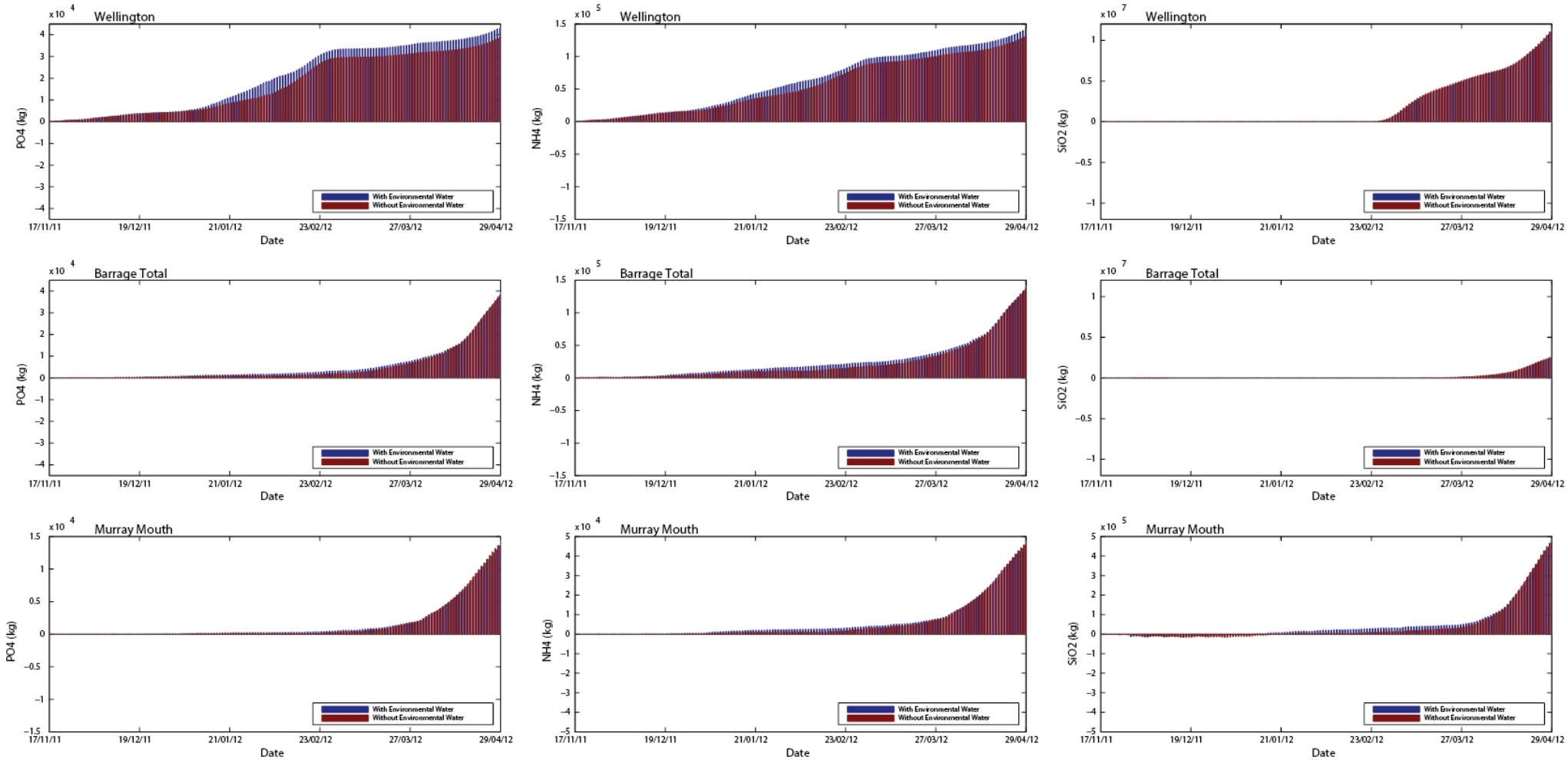


Figure 22. Modelled phosphate (PO_4) ammonium (NH_4) and silica (SiO_2) exports with and without environmental flow provisions. Positive loads are exports passed a designated boundary, including Wellington (from the River Murray to Lake Alexandrina); Barrage Total (from Lower Lakes to Coorong); Murray Mouth (from the Murray Mouth to the Southern Ocean).

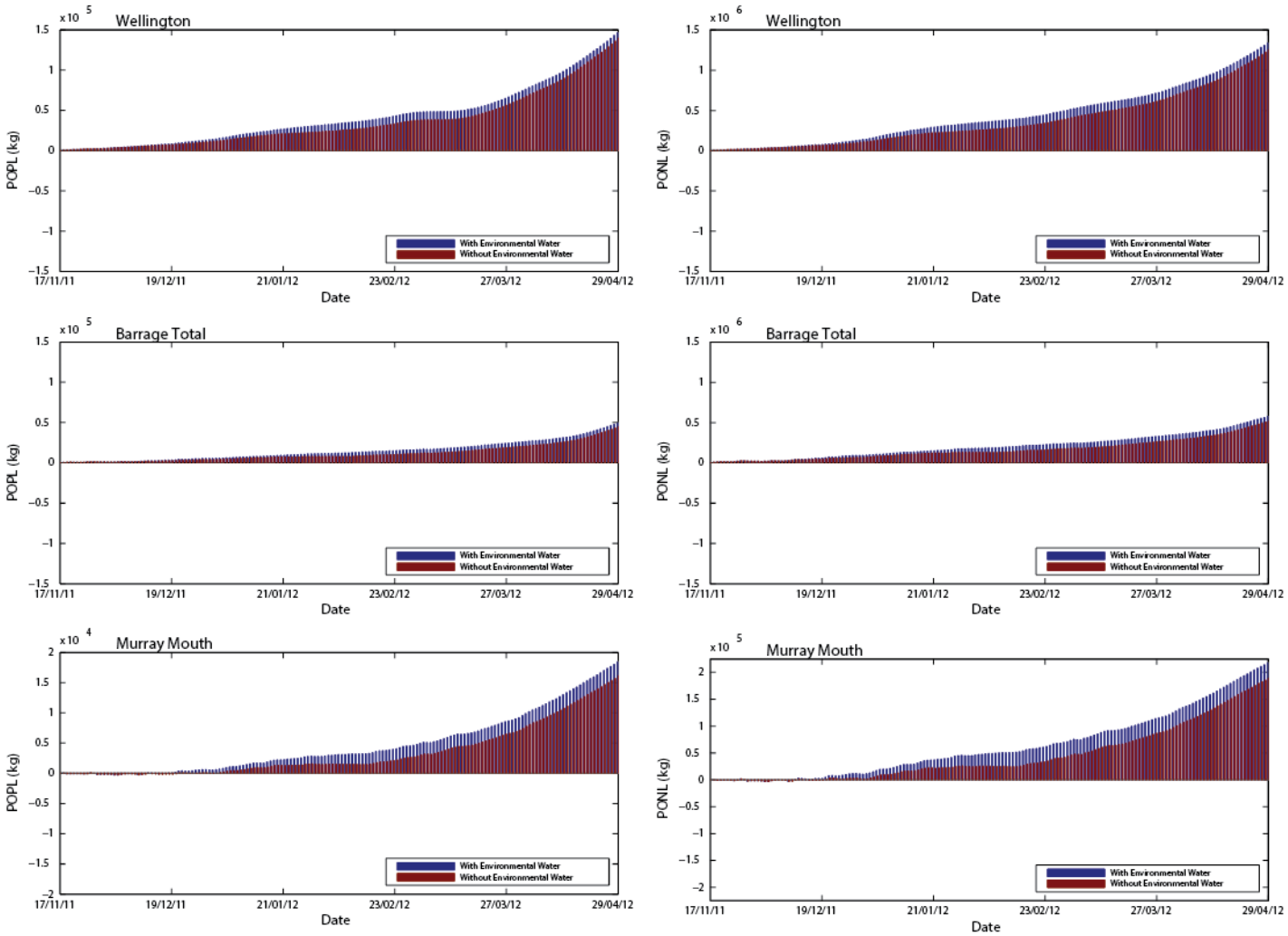


Figure 23. Modelled particulate organic phosphorus (POPL) and nitrogen (PONL) exports with and without environmental flow provisions. Positive loads are exports passed a designated boundary, including Wellington (from the River Murray to Lake Alexandrina); Barrage Total (from Lower Lakes to Coorong); Murray Mouth (from the Murray Mouth to the Southern Ocean).

4 DISCUSSION

4.1 Hydraulic diversity

Whilst total flow volumes, timing and floodplain inundation are important parameters and outcomes of environmental water delivery, within-channel patterns and processes, such as spatio-temporal variability in stream hydraulics, must also be considered. Indeed within-channel hydraulic complexity supports high levels of biodiversity (Statzner and Higler 1986; de Nooij *et al.* 2006), and may influence life history processes in fish (e.g. spawning) (Moir and Pasternack 2008). The current study aimed to characterise the hydraulic complexity of the lower River Murray in two reaches (Lock 1–2 and Lock 5–6), under entitlement flow and a flow of ~15,000 ML d^{-1} as a result of Commonwealth environmental water delivery. Nonetheless, higher volume flows were experienced throughout the study period, due to a combination of natural inflows and environmental water delivery. Thus, we investigated variability in hydraulic complexity at flows of ~13,000–16,000 ML d^{-1} and 23,000–33,000 ML d^{-1}. Additionally, these data were used to 'ground truth' existing predictive hydraulic models developed by Water Technology and DEWNR. Ultimately, these data will inform future environmental flow delivery in the lower River Murray in regards to likely changes in hydraulic diversity under different flow scenarios.

4.1.1 Variability in hydraulic complexity under variable flow

In general, as discharge increased, mean cross-transect velocities increased, velocity range increased, and often, but not in all cases, variability in cross-transect velocities and circulation also increased, indicating greater hydraulic complexity at flows of 23,000–33,000 ML d^{-1} relative to flows of ~13,000–16,000 ML d^{-1}. Whilst in the current study we were unable to characterise hydraulic complexity at low or entitlement flows, water velocities were measured by ADCP in the Lock 4–5 reach in 2006 and 2007 by Kilsby (2008), during very low flows of 3,150–3,670 ML d^{-1} and 724–1,345 ML d^{-1}, respectively. Compared to the current

flow events monitored, low flows resulted in hydraulically homogenous cross-sections, narrow velocity ranges and low mean depth-averaged velocities. At a flow of 3,150 ML d^{-1}, depth averaged velocity ranged 0.05–0.33 $m.s^{-1}$, with a mean of 0.014 $m.s^{-1}$ downstream of Lock 5, compared to a range of -0.01–0.80 $m.s^{-1}$ and a mean of 0.39 $m.s^{-1}$ downstream of Lock 6 at a flow of 13,343 ML d^{-1} in the current study. Kilsby (2008) also found little difference in these parameters between flows of 3,150–3,670 ML d^{-1} and 724–1,345 ML d^{-1} in the Lock 4–5 reach indicating that variability in flow below 4,000 ML d^{-1} produces little variation in hydraulic conditions.

4.1.2 Ground-truthing existing hydraulic models

Mean water velocities calculated from modelled outputs by both Water Technology and DEWNR were generally similar to measured mean velocities, except during higher flows. Mean velocities modelled for the Lock 5–6 reach at flows (QSA) of 13,000, 15,000, 20,000 and 24,000 ML d^{-1} by Water Technology and DEWNR appear accurate when compared to actual mean velocities measured at 16,214 ML d^{-1} (QSA). The Water Technology modelled mean velocity for the upper weir pool location at 40,000 ML d^{-1} (QSA), however, was considerably less than the actual mean velocity at QSA 33,241 ML d^{-1}. This could in part be due to the complex nature of flow provisioning between the River Murray and Chowilla Anabranch system or potentially greater inundation at 40,000 ML d^{-1} and its subsequent impact on within-channel velocities. In the Lock 1–2 reach at flows of ~13,000 ML d^{-1}, DEWNR modelled and actual mean velocities were similar but at flows of ~24,000 ML d^{-1} the modelling underestimated mean velocities across the reach and particularly in the upper weir pool location. These results highlight potential errors in the current hydraulic models at discharges of >20,000 ML d^{-1}.

4.2 Golden perch larvae and larval fish assemblage

Flow regulation has a profound impact on riverine ecosystem processes and associated biological responses. Changes to the natural flow regime, as a direct result of anthropogenic impacts, have been implicated worldwide in the decline of riverine fish communities (e.g. Walker *et al.* 1995; De Leeuw *et al.* 2007). Such ecological impacts are severe throughout the MDB (Gehrke *et al.* 1995), particularly in the lower River Murray (Lloyd and Walker 1986). In recent years, there has been a growing interest in environmental water provision for ecological restoration in riverine systems (Arthington *et al.* 2006). Given fish are recognised as an important ecological asset and provide effective indicators of ecosystem health (Davies *et al.* 2008), they are often targeted for environmental watering actions with specific requirements of hydrological variability considered in flow management (Baumgartner *et al.* 2013). The influence on the success of fish spawning and recruitment are most considered, as these life history processes are critical in maintaining the viability of populations (Trippel and Chambers 1997). This study primarily aimed to investigate the spawning response of golden perch, a flow-cued spawning species, to environmental flow releases to the lower River Murray during spring and summer in 2011-12. In addition, larval fish assemblages in 2011-12 were compared with previous years' data (2005-06 to 2010-11) collected under different flow conditions in the lower River Murray to assess how assemblage structure changes during this within-channel high flow year (2011-12) with releases of Commonwealth environmental water to enhance flow pulses; and if the changes correlate to key environmental variables (i.e. water discharge, relative water level, temperature, conductivity).

4.2.1 Golden perch larvae

Golden perch are known to spawn during spring and summer (Battaglene and Prokop 1987) with extended spawning periods (autumn, winter and spring) also possible (Ebner *et al.* 2009). As flow-

cued spawning species, golden perch and silver perch are the only two native fish species in the MDB that are considered to require increased flow to initiate spawning (Humphries *et al.* 1999); significant recruitment of golden perch typically corresponds to years of increased within-channel flows or overbank floods in the lower River Murray (Ye *et al.* 2008; Leigh *et al.* 2010; Zampatti and Leigh 2013a). Recent (2004-2010) investigations in the lower River Murray also suggest that flows below 10,000 ML d^{-1} are insufficient to induce spawning or support recruitment of golden perch (Ye *et al.* 2008; Leigh *et al.* 2010; Cheshire *et al.* 2012; Zampatti and Leigh 2013a). Given early stage larvae were only collected during flood (2010-11) and within-channel high flow (2011-12) years this study provides further support for the notion that in the lower River Murray, golden perch is a flow-cued spawning species. Collecting early stage larvae (i.e. pre-flexion) that are probably <8 days old, in late summer supports the notion that these fish were spawned recently and likely in the lower River Murray.

Whilst in 2010-11 the substantial increases in discharge and overbank flow conditions seem to have triggered a significant spawning event in late spring, no further larvae were observed later during summer. This may suggest either very low larval abundances that were not detected due to a dilution effect by high overbank flows during summer (>50,000 ML d^{-1}) or the absence of an extended spawning period during the flood. In contrast, in 2011-12, a protracted spawning season (i.e. spring to late summer) was identified for golden perch under within-channel flow conditions with spring/summer discharges ranging between 8,200 and 37,000 ML d^{-1} (at the South Australian border) (Figure 9). A histological study found that female golden perch shed all eggs in a single spawning, and in the absence of suitable stimuli, they would undergo ovarian involution and resorb the eggs (Mackay 1973). However, it was also suggested that multiple spawning events could occur under favourable conditions (Battaglene and Prokop 1987) and may have occured during 2011-12.

In spring 2011-12, there was a major spawning event that seemed to have occurred during the falling phase of the flow pulse coinciding with a temperature rise to >18 °C (Figure 9). In 2012, additional early stage larvae were collected between January and early March, suggesting an extended period of reproductive activities and/or larval drifting from nearby upstream areas. These were likely associated with a summer flow pulse, peaking at ~26,000 ML d^{-1} (at the South Australian border) in early January, and a subsequent increase in discharge from ~15,000 to 25,000 ML d^{-1} (at the South Australian border) in February. These variable within-channel flows may have provided the appropriate conditions to trigger spawning in golden perch, therefore extending its reproductive season well into late summer as hypothesised. The summer flow pulse was enhanced by substantial releases of Commonwealth environmental water in December and January to the lower River Murray. The use of environmental water also helped to slow the rate of recession and maintain discharge at >15,000 ML d^{-1} throughout January, which might have supported the survival and dispersion of golden perch larvae and contributed to recruitment, as suggested by previous studies in the lower River Murray (Cheshire and Ye 2008; Ye *et al.* 2008). A recent study suggested that the elevated flow conditions (>15,000 ML d^{-1}) would favour the productivity of diatoms rather than cyanobacteria and thus better support the aquatic food web (Aldridge *et al.* 2012), including larval fish food resources that are critical for larval survival, growth and subsequent recruitment success.

This study suggests that within-channel flows with natural or environmental flow pulses at appropriate temporal scales may have the potential for generating conditions that favour a protracted reproductive season in this species. The timing of the flow pulses may be of particular relevance to fish spawning, not only because they need to coincide with temperature thresholds but also due to the recovery time required for gonad development between spawning

events. Previous studies suggest that flooding may not always coincide with warm temperatures, and fish are therefore unable to exploit high zooplankton abundances that characterise flood conditions (Humphries *et al.* 1999).

Nutrients, dissolved organic carbon concentrations and phytoplankton abundances in the lower Murray region can be largely driven by upstream inputs (Aldridge *et al.* 2012). During high flow years water from the Darling River appeared to have a disproportionate effect on water quality in the lower Murray (Aldridge *et al.* 2012). Therefore, the origin of the water utilised for environmental watering may also be of importance. Recent studies also show that recruitment of golden perch in the lower River Murray in 2010-11 may have been driven in part by fish spawned in the Darling River (Zampatti and Leigh 2013a). The physical-chemical and biological properties of water likely differ according to their origin therefore changing the potential impact that delivery of water may have on the entire community including the larval fish assemblage.

4.2.2 Larval fish assemblages and flow effects

During 2011-12, a total of ten species were collected as larvae in the main channel of the lower River Murray, comprising nine native and one exotic species. Native species richness in the floodplain region (Site 2) was comparable to the previous flood year (2010-11) and 2005-06, when there was a small flow pulse up to ~15,000 ML d^{-1}. In the gorge region (Site 1), native species richness was the greatest in 2011-12. Increased species richness in the gorge section in 2011-12 was due to the presence of both flow-cued spawning species (golden perch and silver perch) and small-bodied species (hardyhead *spp*.). During 2011-12, the number of carp collected was substantially lower than in 2010-11 (flood year), and no redfin perch were sampled; as carp is a flood opportunistic spawner (King *et al.* 2003), their spawning activity would have increased during the extensive overbank flood in 2010-11.

Prior to 2010, the small to medium-bodied native species (i.e. Australian smelt, bony herring, carp gudgeon and flathead gudgeon) were the most abundant under the low flow and drought conditions (2005-06 to 2008-09). In 2010-11 and 2011-12, there was an environmental change following high flows and extensive flooding; abundance of small to medium-bodied native species decreased whilst larval golden perch were abundant being sampled for the first time since 2005-06. Golden perch and silver perch larvae were only collected during the flood (2010-11), high flow (2011-12) or a small within-channel flow (2005-06), noting that 2005-06 larvae were only collected below Lock 6 (Site 2) in later stages which suggests they may not have been spawned within the lower River Murray in South Australia.

Murray cod and freshwater catfish larvae were collected in greater abundances and distribution during the within-channel flow pulse and high flow/flood years. Both species are known to utilise nests, exhibit parental care of eggs and early stages of larval development (Lake 1967; Allen *et al.* 2002) and spawn annually, irrespective of changes in river flow or level (Davis 1977a, b; Humphries *et al.* 2002; King *et al*. 2003; Gilligan and Schiller 2004; Humphries 2005; Koehn and Harrington 2006, King *et al.* 2009). Despite annual spawning events, years of higher flows and overbank floods in the lower River Murray have been correlated with enhanced recruitment of Murray cod (see Ye and Zampatti 2007). It has been suggested that hydraulically diverse habitats may increase the survival of larval Murray cod and therefore recruitment (Humphries 2005). More recently, environmental flow delivery has been recommended to maximise breeding opportunities for Murray cod by inundating a large number of spawning and nesting sites on a regular basis and to improve connectivity between main channel and off-channel nursery habitat and facilitate movement and dispersion (Baumgartner *et al.* 2013).

The significant change in larval fish assemblage structure in 2011-12 (within-channel high flow), compared to 2005-06 (within-channel low flow) and 2006-07 to 2008-09 (drought years), was mainly due to the presence and increased abundance of the flow-cued spawning species (i.e. golden perch, silver perch) and a reduction in the relative abundance of small to medium-bodied species. The assemblage was more similar to that of the flood year (2010-11), although small-bodied fish larvae were more abundant and carp abundance was lower during the within-channel flows of 2011-12. The results support the hypothesis in regard to the shift of the larval fish assemblage subsequent to the high flows/flood. This shift in larval assemblage reflects the spawning response of large-bodied flow-cued spawning species such as golden perch and silver perch, and also the diminished spawning and/or survival of larvae from small to medium-bodied species that dominated the assemblages in low flow and drought years (i.e. 2005-06 to 2008-09).

The reproduction (spawning and recruitment) of small to medium-bodied fish (carp gudgeon, flathead gudgeon, Australian smelt and bony herring) generally conform with the low flow recruitment hypothesis (LFRH) developed by Humphries *et al.* (1999). These species exhibit an opportunistic life history strategy and are generally resilient to prolonged low flow conditions (Gilligan *et al.* 2009; Cheshire 2010). Studies suggest that under the low flow conditions, the combination of lower velocities and increased prey availability in the river channel may provide suitable conditions for the growth and survival of larvae for small-bodied species (Gehrke 1992; King *et al.* 2003; King 2005; Zeug and Miller 2008). In this study, there was a significant decline in abundances of small to medium-bodied fish during the flood (2010-11) to between 3-20% of that of previous years, suggesting a reduced reproductive performance. A recent fish assemblage study indicates that changes in habitat (a loss of submerged macrophytes) due to increased flow/flood render the main channel of the lower River Murray

less desirable to small-bodied species compared to low-flow periods (Bice *et al*. 2014), supporting the results from the current study. However, in 2011-12 under within-channel flows (up to 26,000 ML d^{-1}), the spawning success of small to medium- bodied species was greater than in the previous flood year, with larval abundance increasing more than sixfold in some cases although still well below those observed in drought years. It is interesting to note that species richness increased in the gorge section of the lower Murray during 2011-12 (when compared to drought years), a within channel high flow year with environmental water releases. This implies that variable within-channel flows may provide an opportunity to meet the different needs of the diverse fish species/guilds. The importance of managing hydrological variability in environmental water use has been demonstrated, considering individual species' contrasting habitat, spawning and feeding requirements and specific life history strategies adapted to different flow regimes. The use of flow guilds of freshwater fish has been suggested as a tool to inform adaptive management of environmental watering in semi-arid riverine systems (Baumgartner *et al*. 2013).

4.3 Recruitment and natal origin of golden perch

In 2012, golden perch populations in the floodplain and gorge geomorphic regions of the lower River Murray were dominated by 1 and 2 year old fish with lower proportions (≤10%) of 0, 6, 11 and 15 year old fish. Overall, our data show episodic recruitment of golden perch during the period of the Millennium drought (2001–2009) but more consistent recruitment from 2009 onwards. Consecutive year-classes from 2010–2012 were spawned in association with in-channel and overbank increases in flow in the lower River Murray and the Darling River, which have improved the resilience and hence health of golden perch populations in the lower River Murray.

Age 0+ (YOY) fish, spawned in spring/summer 2011-12, represented 5% and 10% of fish collected in the gorge and floodplain geomorphic

regions, respectively. Otolith microstructure and geochemistry indicate that these YOY recruits originated from two distinct spawning periods in two different geographic locations. Of the fish for which we calculated approximate spawn dates and have corresponding otolith core Sr isotope ratios (*n* = 15), five fish were most likely spawned in the River Murray (four from 16/10/2011–13/11/2011, and one on 20/12/11) and ten fish spawned in the Darling River from the 27/11/2011–1/1/2012. For the remainder of YOY fish (no specific spawn date) 12 individuals (35%) were spawned in the Murray and 22 (65%) were spawned in the Darling.

Life history profiles of otolith $^{87}Sr/^{86}Sr$ for a 52 and 66 day old golden perch spawned in the Darling indicated that both fish potentially spent over half their lives in the Darling before moving into the River Murray. Given an approximate transit time of 13 days for within-channel flows from Menindee Lakes on the Darling River to the River Murray (Crabb 1997) there is potential that golden perch collected as YOY in the lower Murray in 2012 were spawned many hundreds of kilometres up the Darling. The exact spawning location, however, is unknown and warrants further investigation, potentially using a mixed elemental and isotopic approach (Walther *et al*. 2008).

Ambient $^{87}Sr/^{86}Sr$ ratios in water are dependent on catchment lithology therefore we expect values in the Darling River to be temporally stable (Douglas *et al*. 1995; Gingele and De Deckker 2005). In the River Murray, however, $^{87}Sr/^{86}Sr$ ratios are temporally variable due to multiple sources of water, this is particularly so in the lower River Murray downstream of the Darling River junction. Understanding the true source and delivery rates of environmental water is imperative to determining fish origin. In 2011-12, environmental water was delivered to the lower River Murray directly from source (e.g. Darling River) or traded from upstream sources (e.g. Goulburn and Campaspe rivers) and delivered from Lake Victoria. Given the lack of clarity in the true

source and delivery rates for all components of River Murray discharge into SA, creating a spatio-temporal isotoptic map of water $^{87}Sr/^{86}Sr$ across the southern MDB during each spawning season of golden perch (approximately October–February) would provide the most reliable template from which to determine potential spawning location (Muhlfeld *et al*. 2012).

Given that a significant proportion of the golden perch population in South Australia may result from spawning in the Darling River (or potentially the mid-upper River Murray) maintaining longitudinal connectivity of flow is vital. This is potentially at odds with current water management practices in the lower River Murray where Lake Victoria is used to store and reregulate flows (including environmental flows) from the Murray and Darling Rivers and to moderate salinity in the lower Murray (MDBC 2002). Disruption of hydrologic connectivity by river regulation is considered to be a primary cause of ecological degradation in rivers worldwide (Kondolf *et al*. 2006). In the lower Murray, operation of Lake Victoria disconnects the river continuum. For example, higher salinity water flowing down the Darling River and into the lower Murray may be diverted and substituted or diluted with Lake Victoria water.

Unfortunately, water that is considered 'poor quality' for consumptive use (i.e. irrigation and domestic supply) may be 'biologically' rich and integral to ecosystem function in the lower River Murray. Furthermore, delivering environmental water in a hydrologically disconnected manner potentially mitigates the restorative value of these flows for aquatic ecosystems in the lower Murray.

4.4 Salt and nutrient transport

The approach used in this study proved to be useful for indicating the general response of salt and nutrient dynamics in the Lower Murray to environmental water, although outputs should not be treated as absolute values. The findings suggest that environmental water

provisions in the Lower Murray can influence processes that are essential for providing habitat and resources for aquatic biota. In particular, provisions of environmental water into the Lower Murray appear to have capacity to influence both salt and nutrient concentrations and transport through the lower River Murray, Lower Lakes and Coorong. The study suggests that the provision of environmental water during the study period contributed significantly to the export of salt and particulate nutrients downstream. There was an apparent increase in the export of salt and particulate organic nutrients to the Southern Ocean associated with environmental water, which were associated with elevated outflows through the Murray Mouth and reduced seawater incursions. It is likely that these predictions underestimate salt and nutrient export since not all environmental water provisions could be accounted for and included in the modelling. Since the mass of exported material is a function of both water discharge and concentrations, the provision of additional flows during the study period would be expected to have increased the mass of exported material.

Whilst there was little influence of environmental water on dissolved nutrient concentrations in the model outputs, the elevated loads of dissolved nutrients from the lower River Murray appeared to result in elevated transformation of dissolved nutrients to particulate organic forms. Consequently, elevated particulate organic nutrient concentrations were observed at the Murray Mouth, due to increased inputs from the Lower Lakes (high concentrations) and reduced inputs from the Southern Ocean (low concentrations). At the same time this acted to reduce salinity levels in the Northern Coorong and Murray Mouth due to increased inputs from the Lower Lakes (low salinity levels) and reduced inputs from the Southern Ocean (high salinity levels). The responses observed during the study are largely a function of the antecedent hydrological conditions and underlying hydrological conditions, including the timing, magnitude and source of water used

for environmental flow. Provisions of environmental water during periods of low inflows, in which salts and nutrients tend to accumulate, are likely to have a larger influence on concentrations than were observed during the current study. However, the influence on exports is likely to be smaller if there are insufficient volumes of water to export material. Conversely, when flow is above threshold levels whereby there are exports of water, salt and nutrient exports will largely be proportional to flow and so increased volumes of environmental flow will increase the exported loads. These threshold flow levels will differ between the lower River Murray, Lower Lakes and Murray Mouth and will be influenced by downstream water levels.

5 MANAGEMENT IMPLICATIONS AND RECOMMENDATIONS

This monitoring project assessed key ecological responses during the delivery of Commonwealth environmental water to the South Australian lower River Murray (including the Lower Lakes and Coorong) in 2011-12. These include (1) hydraulic diversity in the river channel, (2) golden perch larvae and larval fish assemblage, (3) recruitment and natal origin of golden perch, and (4) salt and nutrient transport. During the prolonged drought from 2001–2010, the ecological communities in the lower River Murray were severely impacted, many native species suffered a substantial decline and habitat degradation occurred throughout the freshwater and estuarine systems. The significant flood and high flows in 2010–2012 have led to signs of ecological improvement. Importantly, responses to environmental watering need to be considered in the context of ecosystem recovery in the lower River Murray over recent years.

There were a number of ecological outcomes associated with the high flows supported by the releases of Commonwealth environmental water to the lower River Murray during 2011-12. These include a significant increase in hydraulic complexity in the main channel with increased discharge (23,000–33,000 ML d^{-1} QSA); an extended presence of golden perch larvae; increased larval fish abundances of large-bodied species (compared to drought/low flow years); increased species diversity and richness of larval fish particularly in the gorge section of the lower River Murray; aiding the downstream dispersal of early life stage golden perch spawned in the Darling and the subsequent recruitment of these fish in the lower Murray; and a significant contribution to the export of salt and particulate nutrients downstream. The management implications and recommendations are provided for the four components of ecological responses.

5.1 Hydraulic diversity

Whilst we were unable to meet the original objectives of the project due to elevated natural flows, investigating variability in hydraulic complexity between entitlement flows (3,000–7,000 ML d^{-1}) and flows of ~15,000 ML d^{-1}, the current study was able to characterise hydraulic complexity at flows between 13,000 and 34,000 ML d^{-1} and will thus inform future environmental flow delivery of similar volumes. Typically, there was an increase in hydraulic complexity in regards to mean flow velocity, velocity ranges, within location variability in flow velocity and circulation, with increased discharge (23,000–33,000 ML d^{-1} QSA). This suggests that a greater diversity of hydraulic microhabitats may be provided at flows of this magnitude, potentially resulting in a greater ecological response. The positive response to such flow pulses was evident by increased abundances of flow-cued spawning species (golden perch and silver perch) during 2011-12. Therefore, environmental flows of >23,000 ML d^{-1} are preferred to flows of 13,000–16,000 ML d^{-1}. Nevertheless, given that some level of hydraulic complexity was present at 13,000–16,000 ML d^{-1} but virtually absent in the Lock 4–5 reach at flows <4,000 ML d^{-1}, as presented by Kilsby (2008), a priority for future monitoring should be the collection of velocity data at flows of 4,000–13,000 ML d^{-1} to determine potential threshold flow rates beyond which hydraulic complexity begins to increase. It should be noted that whilst within-channel flows at the magnitude of ~15,000 ML d^{-1} may initiate local spawning of flow-cued spawning species (e.g. golden perch), they are also likely to facilitate larval dispersion (i.e. drift) from upstream and thus, enhance recruitment of golden perch in the lower River Murray (Ye *et al.* 2008).

Of considerable importance for future research is developing an understanding of the association of native fish species with hydraulic habitats (at biologically relevant scales) and the explicit link between hydraulics and ecological processes (e.g. spawning and recruitment). Golden perch and silver perch are cued to spawn by increases in

within-channel flows and floods (Lake 1967; Mallen-Cooper and Stuart 2003). Given that variability in hydraulics is how fish perceive changes in flow volumes, in concert with other cues such as temperature (Lake 1967), specific hydraulic conditions may provide the cue for reproductive activity in these species. Additionally, these species together with Murray cod, undergo downstream larval drift (Humphries 2005; Tonkin *et al.* 2007), likely for the purpose of dispersal (reducing competition and predation) and potentially compensation for upstream adult spawning migrations. Thus, hydraulic conditions that facilitate downstream drift and transportation to favourable nursery habitats are likely integral to the recruitment success of these species. Understanding the causal link between hydraulics and vital life history events, and determining the best hydraulic metrics to do so, will provide a powerful tool to inform future environmental water delivery. Ultimately, environmental water requirements could be specified as the provision of particular hydraulic conditions and the flow, and delivery strategies, required to create such conditions rather than a sole reliance on hydrological metrics.

5.2 Golden perch larvae and larval fish assemblage

During 2011-12, summer flow pulses of ~26,000 ML d^{-1} (at the South Australian border) led to an extended reproductive period of golden perch into late summer in the lower River Murray. Importantly, a substantial volume of environmental water was released to enhance the summer flow pulses and reduce the rate of recession in the lower River Murray, which may have contributed to the ecological outcomes of an extended reproductive season and potentially improved larval survival in golden perch. The investigation of golden perch spawning suggests that this species has a flexible reproductive strategy; multiple spawning events could occur when suitable flow conditions were provided. In addition, flow regimes supported by environmental water releases in 2011-12 may have provided suitable conditions to support

spawning and larval survival of fish species from different flow guilds (e.g. flow-cued spawning species, low flow spawners).

Different fish species/guilds often have different requirements of flow and environmental conditions associated with their life history strategies. Fish spawning and larval fish assemblage structure were strongly influenced by hydrological and environmental conditions in the lower River Murray. Three types of larval fish responses were evident: species whose larval abundances were 1) positively correlated to increased discharge, 2) negatively correlated to the increased discharge and 3) correlated to temperature. In this study, small to medium-bodied native species (e.g. Australian smelt, flathead gudgeon, carp gudgeon and bony herring) were related to low flows; whereas, the large-bodied Murray cod, golden perch, silver perch and freshwater catfish were considered to be related to high flows (e.g. overbank and within-channel flows). The correlation between temperature and changes in larval fish assemblage most likely reflects within season differences in spawning of individual species (King *et al.* 2003; Cheshire 2010).

Under prolonged drought conditions, extended periods of low flow and floodplain isolation have negatively impacted on the spawning and recruitment of some of the large-bodied species (Ye *et al.* 2008; King *et al.* 2009; Cheshire 2010; Leigh and Zampatti 2011). Although large-bodied, longer-lived species are likely to be able to withstand unfavourable spawning and recruitment conditions for longer, many of these species are adapted to hydrological variability, which has strong influence on their life history processes (e.g. spawning, larval survival, movement and migration). Subsequently, they are more threatened by river regulation and water extraction. In addition these species tend to be slower at recovering from population decreases (Baker *et al.* 2009). This study has highlighted that the abundance and distribution of Murray cod and freshwater catfish larvae was greater during higher

flow conditions; and the two flow-cued spawning species, golden perch and silver perch, only spawned in conjunction with overbank or increased within-channel flows in the lower River Murray. Therefore, these species are more likely to be under threat during extended low flow conditions (<10,000 ML d^{-1} at the South Australian border). Environmental flow allocation and management should take into account the requirements of these native species in order to improve the resilience of the populations.

Whilst floods are important in maintaining the ecological integrity of floodplain rivers, this study suggests that within-channel flow management may present an opportunity to enhance the populations of golden perch and other native fish. In the lower River Murray, within-channel flows (e.g. 15,000–50,000 ML d^{-1}) are currently the main focus for environmental flow management, which could be restored within the current constraints of system operation. If environmental flows can be delivered in an appropriate regime (magnitude, timing, duration, frequency) at relevant biological/ecological scales, they may facilitate the spawning and recruitment of flow-cued spawning species and other large-bodied native fish. Furthermore, within-channel flows should also provide suitable conditions for the reproduction of small to medium-bodied species, resulting in a diverse fish community. Finally, given there are still significant knowledge gaps in environmental flow science with regard to the restoration of native fish populations (King *et al.* 2009), ecological research and adaptive monitoring are essential to underpin environmental flow management.

5.3 Recruitment and natal origin of golden perch

Resilient populations of long-lived native fish necessitate multiple age classes to maximise population viability in light of environmental perturbations and anthropogenic impacts (Gunderson 2000). Consequently, assessment of population resilience requires an understanding of survivorship and population demographics. To date

in the southern MDB there has been an emphasis on short-term monitoring of ecological responses to environmental water allocations (CEWO 2012) that may not provide data on population structure. For long-lived native fish, longer-term investigations of population demographics are required to establish the status (i.e. health) of populations and the true benefits of flow restoration.

The results of the present study support the hypothesis that flows >15,000 ML d^{-1} during summer 2011-12 would result in recruitment (to age 0+) of golden perch in the South Australian reaches of the River Murray but that these YOY fish would not necessarily have been spawned in the lower River Murray. Whilst we cannot determine a precise spawning location for the 35% of YOY fish spawned in the River Murray, we have ascertained that 65% of YOY were spawned in the Darling River. Furthermore, when age is also considered, the sample of larval and YOY golden perch collected in the present study, that were recruited to the lower River Murray population in 2011-12, were generally not spawned in association with the delivery of Commonwealth environmental water to the lower Murray channel. Nevertheless, substantial volumes (4,000–5,000 ML d^{-1}) of Commonwealth environmental water delivered down the Darling from January–February 2012 would have aided the downstream dispersal of early life stage golden perch spawned in the Darling and the subsequent recruitment of these fish in the lower Murray.

5.4 Salt and nutrient transport: Lock 1 to the Southern Ocean

Continued refinement of the model used in this study will further improve its capacity to assess the response of salt and nutrient dynamics in the Lower Murray to environmental water. In the future it could also be used to assess the various watering actions in both forecasting and retrospective analysis. Without such assessments it is difficult to reach general recommendations about optimal use of environmental water for salt and nutrient dynamics because the Lower

Murray is a hydrologically complex system. However, based on insights provided by this study and knowledge of nutrient dynamics in the Lower Murray, the following points could be used to help guide future environmental water provisions:

- Environmental flow provisions during moderate-high flow periods (e.g. >40,000 ML d^{-1}) are likely to have greater impacts on salt and nutrient exports from the Murray Mouth than during low flow periods when the Murray Mouth outflows are small. In contrast, environmental flow provisions during low flow periods (e.g. <10,000 ML d^{-1}) are likely to have greater impacts on salt and nutrient concentrations than during moderate-high flow periods.
- Maximum exports from the Murray Mouth are likely to be achieved by delivering environmental water during periods of low oceanic water levels (summer). However, whilst this may have short-term benefits, reduced water provisions at other times are likely to increase the import of material from the Southern Ocean. In contrast, delivery of environmental water to the region at times of high oceanic water levels is likely to increase the exchange of water and associated nutrients and salt through the Coorong, rather than predominately through the Murray Mouth.
- Environmental water provision that results in floodplain inundation will likely result in increased nutrient concentrations (mobilisation) and export. This may be achieved by moderate-large floods (e.g. >40,000 ML d^{-1}) that inundate previously dry floodplains and wetlands.
- Flows during winter may result in limited assimilation of nutrients by biota (slower growth rates), whilst provisions during summer could increase the risk of blackwater events and cyanobacterial blooms, depending on hydrological conditions and the degree of wetland connectivity. Flows during spring are likely to minimise these risks, but also maximise the benefits of nutrient inputs (e.g. stimulate productivity to support larval survival).

- Multiple watering events in a given year could be used to meet different objectives. For example providing one event in spring to increase nutrient assimilation, followed by a subsequent event to export material from the system may provide multiple benefits.

REFERENCES

Agostinho, A.A., Gomes, L.C., Verissimo, S. and Okada, E.K. (2004) Flood regime, dam regulation and fish in the upper Parana River: Effects on assemblage attributes, reproduction and recruitment. *Reviews in Fish Biology and Fisheries* **14(1)**, 11-19.

Aldridge, K.T., Lamontagne, S., Deegan, B.M. and Brookes, J.D. (2011) Impact of a drought on nutrient concentrations in the Lower Lakes (Murray–Darling Basin, Australia). *Inland Waters* **1**, 159-176.

Aldridge, K., Lorenz, Z., Oliver, R. and Brooks, J.D. (2012) Changes in water quality and phytoplankton communities in the lower River Murray in response to a low flow-high flow sequence. Goyder Institute for Water Research. Adelaide, Australia

Aldridge, K.T., Busch, B.D. and Hipsey, M.R. (2013) An assessment of the contribution of environmental water provisions to salt and nutrient dynamics in the lower Murray, November 2011-July 2012: Technical report. The University of Adelaide and the University of Western Australia, Adelaide and Perth.

Allen, G.R., Midgley, S.H. and Allen, M. (2002) Field guide to the freshwater fishes of Australia. (Western Australian Museum: Perth)

Anderson, J. R., Morison, A. K. and Ray, D. J. (1992). Validation of the use of thin-sectioned otoliths for determining age and growth of golden perch, *Macquaria ambigua* (Perciformes: Percichthyidae), in the Lower Murray–Darling Basin, Australia. *Australian Journal of Marine and Freshwater Research* **43**, 1103–1128.

Arthington, A.H., Bunn, S.E., Poff, L.N. and Naiman , R.J. (2006) The challenge of providing environmental flow rules to sustain river ecosystems. *Ecological Applications* **16**, 1311-1318.

Baker, K.D., Devine, J.A. and Haedrich, R.L. (2009) Deep-sea fishes in Canada's Atlantic: Population declines and predicted recovery times. *Environmental Biology of Fishes* **85(1)**, 79-88.

Baumgartner, L.J., Conallin, J., Wooden, I., Campbell B., Gee, R., Robinson, W.A. and Mallen-Cooper, M. (2013) Using flow guilds of freshwater fish in an adaptive management framework to simplify environmental flow delivery for semi-arid riverine systems. *Fish and Fisheries*. DOI: 10.1111/faf.12023

Battaglene, S. and Prokop, F. (1987) Golden perch. N.S.W. Department of Agriculture No. A3.2.2, Sydney.

Bertozzi, T., Adams, M. and Walker, K.F. (2000) Species boundaries in carp gudgeons (Eleotrididae : *Hypseleotris*) from the River Murray, South Australian: Evidence for multiple species and extensive hybridization. *Marine and Freshwater Research* **51(8)**, 805-815.

Bice, C. M., Leigh, S. J., Nicol, J. M. and Zampatti, B. P. (2013) Changes in hydraulic complexity in the lower River Murray main channel in relation to flow variability. South Australian Research and Development Institute (Aquatic Sciences), Adelaide. SARDI Publication No. F2013/000214-1. SARDI Research Report Series No. 709. 32pp.

Bice, C. M., Gehrig S. L., Zampatti B. P., Nicol J. M., Wilson P., Leigh S. L. and Marsland K. (2014). Flow-induced alterations to fish assemblages, habitat and fish–habitat associations in a regulated lowland river. *Hydrobiologia* **722**: 205-222.

Blanch, S. J., Walker, K. F. and Ganf, G. G. (2000) Water regimes and littoral plants in four weir pools of the River Murray, Australia. *Regulated Rivers: Research & Mangement* **16**, 445-456.

Brookes, J.D., Lamontagne, S., Aldridge, K.T., Benger, S., Bissett, A., Bucater, L., Cheshire, A.C., Cook, P.L.M., Deegan, B.M., Dittmann, S., Fairweather, P.G., Fernandes, M.B., Ford, P.W., Geddes, M.C., Gillanders, B.M., Grigg, N.J., Haese, R.R., Krull, E., Langley, R.A., Lester, R.E., Loo, M., Munro, A.R., Noell, C.J., Nayar, S., Paton, D.C., Revill, A.T., Rogers, D.J., Rolston, A., Sharma, S.K., Short, D.A., Tanner, J.E., Webster, I.T., Wellman, N.R. and Ye, Q. (2009) An ecosystem assessment framework to guide management of the Coorong. CSIRO: Water for a Healthy Country National Research Flagship, Canberra.

Brown, A. V. and Armstrong, M. L. (1985) Propensity to drift downstream among various species of fish. *Journal of Freshwater Ecology* **3,** 3-17.

Brown, P. and Wooden, I. (2007). Age at first increment formation and validation of daily growth increments in golden perch (*Macquaria ambigua*: Percicthyidae) otoliths. *New Zealand Journal of Marine and Freshwater Research* **41**, 157–161.

Bucater, L., Cheshire, K. and Ye, Q. (2009) Fish spawning in the lower River Murray icon sites, South Australia: with reference to drought intervention. South Australian Research and Development Institute (Aquatic Sciences), No. F2009/000360-1, Adelaide.

Bunn, S. and Arthington, A.H. (2002) Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. *Environmental management* **30**, 492-507.

Cambray, J.A., King, J.M. and Bruwer, C. (1997) Spawning behaviour and early development of the Clanwilliam yellowfish (*Barbus capensis*; Cyprinidae), linked to experimental dam releases in the Olifants River, South Africa. *Regulated Rivers-Research & Management* **13(6)**, 579-602.

Campana, S. E. (1999) Chemistry and composition of fish otoliths: pathways, mechanisms and applications. *Marine Ecology Progress Series* **188**, 263–297.

Commonwealth Environmental Water Office (2012) 2011 – 2012 Outcomes Report. Commonwealth of Australia, Canberra.

Cheshire, K.J.M. and Ye, Q. (2008) Spatial and temporal variations in larval fish assemblages between Locks 1 and 6 in the River Murray, South Australia: with reference to drought intervention monitoring 2007. SARDI Aquatic Sciences Publication No. F2008/000685-1. SARDI Research Report Series No 304.

Cheshire, K.J.M. (2010) Larval fish assemblages in the lower River Murray, Australia: examining the influence of hydrology, habitat and food. Ph.D. Thesis. The University of Adelaide. Australia.

Cheshire, K.J.M., Ye, Q., Wilson, P. and Bucater, L. (2012) From drought to flood: Annual variation in larval fish assemblages in larval fish assemblages in a heavily regulated lowland temperate river. Technical Report Series No.12/6, Goyder Institute for Water Research, Adelaide.

Cook, P.L.M., Aldridge, K.T., Lamontagne, S. and Brookes, J.D. (2010) Retention of nitrogen phosphorus and silicon in a large semi-arid riverine lake system. *Biogeochemistry* **99 (1-3)**, 49-63.

Crabb, P. (1997) Murray–Darling Basin Resources. Murray–Darling Basin Commission, Canberra.

Crowder, D. W. and Diplas, P. (2000) Evaluating spatially explicit metrics of stream energy gradients using hydrodynamic model simulations. *Canadian Journal of Fisheries and Aquatic Sciences* **57,** 1497-1507.

CSIRO (2008) Water availability in the Murray–Darling Basin. Report to the Australian government from the CSIRO Murray–Darling Basin Sustainable Yields Project. CSIRO, Australia, 67pp.

Davies, .PE., Harris, J.H., Hillman, T.J. and Walker, K.F. (2008) Sustainable Rivers Audit Report 1: A Report on the Ecological Health of Rivers in the Murray–Darling Basin, 2004–2007. Prepared by the Independent Sustainable Rivers Audit Group for the Murray-Darling Basin Ministerial Council.

Davis, T.L.O. (1977a) Food habits of the freshwater catfish, *Tandanus tandanus* Mitchell, in the Gwydir River, Australia, and effects associated with impoundment of this river by the Copleton Dam. *Australian Journal of Marine and Freshwater Research* **28(4)**, 455-465.

Davis, T.L.O. (1977b) Reproductive biology of the freshwater catfish, *Tandanus tandanus* Mitchell, in the Gwydir River, Australia i. Structure of the gonads. *Australian Journal of Marine and Freshwater Research* **28**, 139-158.

De Leeuw, J.J., Buijse, A.D., Haidvogl, G., Lapinska, M., Noble, R., Repecka, R., Virbickas, T., Wisniewolski, W. and Wolter, C. (2007) Challenges in developing fish-based ecological assessment methods for large floodplain rivers. *Fisheries Management and Ecology* **14(6)**, 483-494.

de Nooij, R. J. W., Verberk, W. C. E. P., Lenders, H. J. R., Leuven, R. S. E. W. and Nienhuis, P. H. (2006) The importance of hydrodynamics for protected and endangered biodiversity of lowland rivers. *Hydrobiologia* **565**, 153-162.

Douglas, G. B, Gray, C. M., Hart, B. T. and Beckett, R. (1995) A strontium isotopic investigation of the origin of suspended particulate matter (SPM) in the Murray-Darling river system, Australia. *Geochimica et Cosmochimica Acta* **59**, 3799–3815.

Dyer, F. J. and Thoms, M. C. (2006) Managing river flows for hydraulic diversity: an example from an upland regulated gravel-bed river. *Regulated Rivers: Research & Mangement* **22**, 257-267.

Ebner, B.C., Scholz, T. and Gawne, B. (2009) Golden perch *Macquaria ambigua* are flexible spawners in the Darling River, Australia. *New Zealand Journal of Marine and Freshwater Research* **43(2)**, 571-578.

Freeman, M. C. and Grossman, G. D. (1993) Effects of habitat availability on dispersion of a stream cyprinid. *Environmental Biology of Fishes* **37**, 121-130.

Freeman, M.C., Bowen, Z.H., Bovee, K.D. and Irwin, E.R. (2001) Flow and habitat effects on juvenile fish abundance in natural and altered flow regimes. *Ecological Applications* **11**, 179-190.

Gehrke, P.C. (1992) Diel abundance, migration and feeding of fish larvae (Eleotridae) in a floodplain billabong. *Journal of Fish Biology* **40(5)**, 695-707.

Gehrke, P.C., Brown, P., Schiller, C.B., Moffatt, D.B. and Bruce, A.M. (1995) River regulation and fish communities in the Murray-Darling river system, Australia. *Regulated Rivers: Research and Management* **11**, 363-375.

Gillanders, B. M. (2005) Otolith chemistry to determine movements of diadromous and freshwater fish. *Aquatic Living Resources* **18**, 291–300.

Gilligan, D. and Schiller, C. (2004) Downstream transport of the eggs, larvae and juvenile fish in the Murray river: Impact of river regulation on downstream dispersal. In 'Downstream movement of fish in the Murray-Darling Basin', 3-4 June 2003, Canberra Workshop. (Eds. M Lintermans and B Phillips), pp. 41-50

Gilligan, D.M., Vey, A. and Asmus, M. (2009) Identifying drought refuges in the Wakool system and assessing status of fish populations and water quality before, during and after the provision of environmental, stock and domestic flows. 58 pp.

Gingele, F. X. and De Decker, P. (2005) Clay mineral, geochemical and Sr–Nd isoptopic fingerprinting of sediments in the Murray–Darling fluvial system, southeast Australia. *Australian Journal of Earth Sciences* **52**, 965–974.

Graham, R. and Harris, J.H. (2004) Floodplain inundation and fish dynamics in the Murray-Darling Basin. Current concepts and future research: a scoping study. Cooperative Centre for Freshwater Ecology. Canberra. 52 pp.

Gunderson, L. H. (2000) Ecological resilience – in theory and application. *Annual Review of Ecology and Systematics* **31**, 425–439.

Hart, B.T., Bailey, P., Edwards, R., Hortle, K., James, K., McMahon, A., Meredith, C. and Swadling, K. (1991) A review of salt sensitivity of the Australian freshwater biota. *Hydrobiologia* **210**, 105-144.

Henderson, A. R. and Johnston, C. E. (2010) Ontogenetic habitat shifts and habitat use in an endangered minnow, *Notropis mekistocholas*. *Ecology of Freshwater Fish* **19**: 87-95.

Hipsey, M.R. and Busch, B.D. (2012) Lower lakes water quality recovery dynamics. University of Western Australia, Perth.

Hobbs, J. A., Yin, Q., Burton, J. and Bennett, W. A. (2005) Retrospective determination of natal habitats for an estuarine fish with otolith strontium isotope ratios. *Marine and Freshwater Research* **56**, 655–660.

Humphries, P., King, A.J. and Koehn, J.D. (1999) Fish flows and flood plains: Links between freshwater fishes and their environment in the Murray-Darling river system, Australia. *Environmental Biology of Fishes* **56**, 129-151.

Humphries, P., Serafini, L.G. and King, A.J. (2002) River regulation and fish larvae: Variation through space and time. *Freshwater Biology* **47(7)**, 1307-1331.

Humphries, P. (2005) Spawning time and early life history of the Murray cod, *Maccullochella peelii peelii* (Mitchell) in an Australian river. *Environmental Biology of Fishes* **72**, 393-407.

Junk, W.J., Bayley, P.B. and Sparks, R.E. (1989) The flood-pulse concept in river-floodplain systems. In Proceedings of the international large river symposium. **106** (Ed. DP Dodge) pp. 110-27. *Canadian Special Publication of Fisheries and Aquatic Sciences*

Kilsby, N. N. (2008) Reach-scale spatial hydraulic diversity in lowland rivers: characterisation, measurement and significance for fish. Ph.D. Thesis. The University of Adelaide.

Kilsby, N. N. and Walker, K. F. (2012) Behaviour of two small pelagic and demersal fish species in diverse hydraulic environments. *River Research and Applications* **28,** 543-553.

King, A.J., Humphries, P. and Lake, P.S. (2003) Fish recruitment on floodplains: The roles of patterns of flooding and life history characteristics. *Canadian Journal of Fisheries and Aquatic Sciences* **60(7)**, 773-786.

King, A.J. (2005) Ontogenetic dietary shifts of fishes in an Australian floodplain river. *Marine and Freshwater Research* **56**, 215-225.

King, A.J., Tonkin, Z. and Mahoney, J. (2009) Environmental flow enhances native fish spawning and recruitment in the Murray River, Australia. *River Research and Applications* **25**, 1205-1218.

Kingsford, R.T. (2000) Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. *Austral Ecology* **25(2)**, 109-127.

Koehn, J.D. and Harrington, D.J. (2006) Environmental conditions and timing for the spawning of murray cod (*Maccullochella peelii peelii*) and the endangered trout cod (*M. macquariensis*) in southeastern Australian rivers. *River Research and Applications* **22(3)**, 327-342.

Kondolf, G. M., Boulton, A. J., O'Daniel, S., Poole, G. C., Rahel, F. J., Stanley, E. H., Wohl, E., Bang, A., Carlstrom, J., Cristtoni, C., Huber, H., Koljonen, S., Louhi, P. and Nakamure, K. (2006) Process-based ecological river restoration: visualizing three-dimensional connectivity and dynamic vectors to recover lost linkages. *Ecology and Society* **11**,

Lake, J.S. (1967) Rearing experiments with five species of Australian freshwater fishes: I. Inducement to spawning. *Australian Journal of Marine and Freshwater Research* **18**, 137-153.

Leigh, S., Zampatti, B. and Nichol, J.M. (2010) Chowilla icon site – fish assemblage condition monitoring 2005 – 2010. South Australian Research and Development Institute (Aquatic Sciences), No. F2008/000907-2.

Leigh, S. and Zampatti, B. (2011) Movement and spawning of Murray cod (*Maccullochella peelii*) and golden perch (*Macquaria ambigua ambigua*) in response to a small-scale flow manipulation in the Chowilla Anabranch. South Australian Research and Development Institute (Aquatic Sciences), No. F2010/000646-1.

Lloyd, L.N. and Walker, K.F. (1986) Distribution and conservation status of small freshwater fish in the River Murray, South Australia. *Transactions of the Royal Society of South Australia* **110(2)**, 49-57.

Lytle, D.A. and Poff, N.L. (2004) Adaptation to natural flow regimes. *Trends in Ecology & Evolution* **19(2)**, 94-100.

Mackay, N.J. (1973) Histological changes in the ovaries of the golden perch, *Plectroplites ambiguus*, associated with the reproductive cycle. *Australian Journal of Marine and Freshwater Research* **24**, 95-101.

Maheshwari, B.L., Walker, K.F. and McMahon, T.A. (1995) Effect of regulation on the flow regime of the River Murray, Australia. *Regulated Rivers: Research and Management* **10**, 15-38.

Mallen-Cooper, M. and Stuart, I. G. (2003) Age, growth and non-flood recruitment of two potamodromous fishes in a large semi-arid/temperate river system. *River Research and Applications* **19**: 697-719.

Mallen-Cooper, M., Zampatti, B., Hillman, T., King, A., Koehn, J., Saddlier, S., Sharpe, S. and Stuart, I.G. (2011) Managing the Chowilla Creek Environmental Regulator for Fish Species at Risk, Report prepared for the South Australian Murray-Darling Basin Natural Resources Management Board. 128 pp.

McArthur, J.M. and Howarth, R.J. (2004). Sr-isotope stratigraphy: the Phanerozoic 87Sr/86Sr-curve and explanatory notes. In 'A geologic timescale 2004'. (Eds. F. M. Gradstein, J.G. Ogg and A. G. Smith) Cambridge University Press, Cambridge, p. 96–105.

Murray–Darling Basin Commission (2002) *Lake Victoria Operating Strategy*. MDBC Technical Report No. 2002/01. Murray–Darling Basin Commission, Canmberra.

MDBC (2004) Native fish strategy for the Murray-Darling basin 2003-2013. Murray–Darling Basin Basin Commision, MDBC Publication No. 25/04, Canberra, ACT.

Moir, H. J. and Pasternack, G. B. (2008) Relationships between mesoscale morphological units, stream hydraulics and Chinook salmon (*Oncorhynchus tshawytscha*) spawning habitat on the Lower Yuba River, California. *Geomorphology* **100**: 527-548.

Molles, M.C., Crawford, C.S., Ellis, L.M., Valett, H.M. and Dahm, C.N. (1998) Managed flooding for riparian ecosystem restoration - managed flooding reorganizes riparian forest ecosystems along the middle Rio Grande in New Mexico. *Bioscience* **48(9)**, 749-756.

Mosley, L.M., Zammit, B., Corkhill, E., Heneker, T.M., Hipsey, M.R., Skinner, D.S. and Aldridge, K.T. (2012) The impact of extreme low flows on the water quality of the lower Murray River and lakes, South Australia. *Water Resources Management* **26**, 3923-3946.

Muhlfeld, C. C., Thorrold, S. R., McMahon, T. E. and Marotz, B. (2012) Estimating westslope cutthroat trout (Oncorhynchus clarkia lewisi) movements in a river network using strontium icoscapes. *Canadian Journal of Fisheries and Aquatic Sciences* **69**, 906–915.

Naiman, R.J., Bunn, S.E., Nilsson, C., Petts, G.E., Pinay, G. and Thompson, L.C. (2002) Legitimizing fluvial ecosystems as users of water: An overview. *Environmental management* **30(4)**, 455-467.

Nestler, J.M., Pompeu, P.S., Goodwin, R.A., Smith, D.L., Silva, L.G.M., Baigun, C.R.M. and Oldani, N.O. (2012) The river machine: A template for fish movement and habitat, fluvial geomorphology, fluid dynamics and biogeochemical cycling. *River Research and Applications* **28**, 490-503.

Nielsen, D.L., Brock, M.A., Rees, G.N. and Baldwin, D.S. (2003) Effects of increasing salinity on freshwater ecosystems in Australia. *Australian Journal of Botany* **51**, 655-665.

Nislow, K. H., Folt, C. L. and Parrish, D. L. (1999) Favorable foraging locations for young Atlantic salmon: application to habitat and population restoration. *Ecological Applications* **9,** 1085-1099.

Palmer, M.R., and Edmond, J.M. (1989) The strontium isotope budget of the modern ocean. *Earth and Planetary Science Letters* **92**, 11-26.

Paton, C., Hellstrom, J., Paul, B., Woodhead, J., and Hergt, J. (2011) Iolite: Freeware for the visualisation and processing of mass spectrometric data. *Journal of Analytical Atomic Spectrometry* **26**, 2508–2518.

Pin, C., Briot, D., Bassin, C. and Poitrasson, F. (1994) Concomitant separation of strontium and samarium-neodymium for isotopic analysis in silicate samples, based on specific extraction chromatography. *Analytica Chimica Acta* **298**, 209–217.

Poff, N.L., Allan, J.D., Bain, M.B., Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E. and Stromberg, J.C. (1997) The natural flow regime: A paradigm for river conservation and restoration. *BioScience* **47(11)**, 769-784.

Poff, N.L. and Zimmerman, J.K.H. (2010) Ecological responses to altered flow regimes: A literature review to inform the science and management of environmental flows. *Freshwater Biology* **55(1)**, 194-205.

Potter, N.J., Chiew, F.H.S. and Frost, A.J. (2010) An assessment of the severity of recent reductions in rainfall and runoff in the Murray-Darling basin. *Journal of Hydrology* **381(1-2)**, 52-64.

Puckridge, J.T., Sheldon, F., Walker, K.F. and Boulton, A.J. (1998) Flow variability and the ecology of large rivers. *Marine and Freshwater Research* **49(1)**, 55-72.

Richter, B.D., Baumgartner, J.V., Powell, J. and Braun, D.P. (1996) A method for assessing hydrologic alteration within ecosystems. *Conservation Biology* **10(4)**, 1163-1174.

Serafini, L.G. and Humphries, P. (2004) Preliminary guide to the identification of larvae of fish, with a bibliography of their studies, from the Murray-Darling basin. Cooperative Research Centre for Freshwater ecology, Murray-Darling Freshwater Research Centre, Albury and Monash University, Clayton, No. Identification and Ecology Guide No. 48.

Shields, F. D. and Rigby, J. R. (2005) River habitat quality from river velocities measured using acoustic doppler current profiler. *Environmental Management* **36,** 565-575.

Statzner, B. and Higler, B. (1986) Stream hydraulics as a major determinant of bethic invertebrate zonation patterns. *Freshwater Biology* **16,** 127-139.

The MathsWorks Inc (2010). MATLAB version 7.10.0. Natick, Massachusetts

Tockner, K. and Stanford, J.A. (2002) Riverine flood plains: Present state and future trends. *Environmental Conservation* **29(3)**, 308-330.

Tonkin, Z. D., Humphries, P. and Pridmore P. A. (2006) Ontogeny of feeding in two native and one alien fish species from the Murray-Darling Basin, Australia. *Environmental Biology of Fishes* **76**, 303–315.

Tonkin, Z., King, A., Mahoney, J. and Morrongiello, J. (2007) Diel and spatial drifting patterns of silver perch *Bidyanus bidyanus* eggs in an Australian lowland river. *Journal of Fish Biology* **70**, 313-317.

Travnichek, V.H., Bain, M.B. and Maceina, M.J. (1995) Recovery of a warmwater fish assemblage after the initiation of a minimum-flow release downstream from a hydroelectric dam. *Transactions of the American Fisheries Society* **124(6)**, 836-844.

Trippel, E.A. and Chambers, R.C. (1997) The early life history of fishes and its role in recruitment processes. In Early life history and recruitment in fish populations. (Eds. RC Chambers and EA Trippel) pp. 21-32. Chapman and Hall, London, UK.

Walker, K.F. (1985) A review of the ecological effects of river regulation in Australia. *Hydrobiologia* **125**, 111-129.

Walker, K. F. and Thoms, M. C. (1993) Environmental effects of flow regulation on the lower River Murray, Australia. *Regulated Rivers: Research & Management* **8,** 103-119

Walker , K.F., Sheldon, F. and Puckridge, J.T. (1995) An ecological perspective on dryland rivers. *Regulated Rivers: Research & Management* **11**, 85-104.

Walker , K.F. (2006) Serial weirs cumulative effects: The lower River Murray, Australia. In Ecology of desert rivers. (Ed. RT Kingsford) pp. 248-279. Cambridge University Press, Cambridge.

Walther, B. D., Thorrold, S. R. and Olney, J. E. (2008) Geochemical signatures in otoliths record natal origins of American shad. *Transactions of the American Fisheries Society* **137**, *57–69*.

Wassens, S., Watts, R.J., Spencer, J.A., Howitt, J., McCasker, N.A., Griese, V., Burns, A., Croft, R., Zander, A., Amos, C. and Hall, A. (2012) Monitoring of ecosystem responses to the delivery of environmental water in the Murrumbidgee system. Institute for Land, Water and Society, Charles Sturt University, Report 2.

Weisberg, S.B. and Burton, W.H. (1993) Spring distribution and abundance of ichthyoplankton in the tidal Delaware River. *Fishery Bulletin* **91(4)**, 788-797.

Welcomme, R.L. (1985) River fisheries. United Nations, No. 262, Rome.

Williams, W.D. (1987) Salinisation of rivers and streams: An important environmental hazard. *Ambio* **16**, 180-185.

Woodhead, J., Swearer, S., Hergt, J. and Maas, R. (2005) In situ Sr-isotope analysis of carbonates by LA-MC-ICP-MS: interference corrections, high spatial resolution and an example from otolith studies. *Journal of Analytic Atomic Spectrometry* **20**, 22–27.

Ye, Q. and Zampatti, B.P. (2007) Murray cod stock status: The Lower River Murray, South Australia. South Australian Research and Development Institute (Aquatic Sciences), No. SARDI publication No. F20007-000211-1, Adelaide, SA.

Ye, Q., Cheshire, K.J. and Fleer, D. (2008) Recruitment of golden perch and selected large-bodied fish species following the weir pool manipulation in the river murray, south australia. South Australian Research and Development Institute (SARDI), Aquatic Sciences, SARDI Research Report Series No. 303, Publication No. F2008/000606-1, Adelaide.

Ye, Q., Livore, J.P., Bucater, L.B., Cheshire, K.J.M. and Fleer, D. (2013) Changes in larval fish assemblages related to Commonwealth environmental watering in the lower River Murray in 2011/12 with emphasis on golden perch. South Australian Research and Development Institute (Aquatic Sciences), SARDI Publication No. F2013/000000-1. Adelaide.

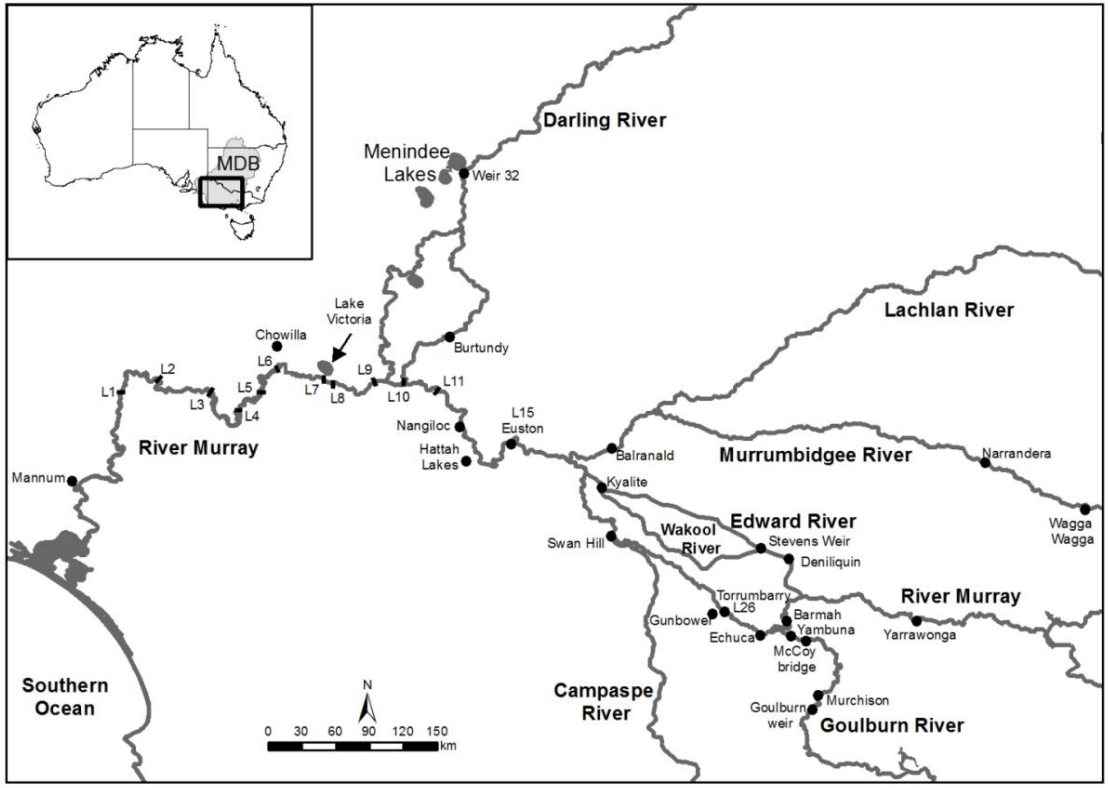
Zampatti, B. P., Leigh, S. J. and Nicol, J. M. (2011) Fish and aquatic macrophyte communities in the Chowilla Anabranch System, South Australia: A report on investigations from 2004 – 2007. South Australia Research and Development Institute (Aquatic Sciences), SARDI Research Report Series No. 525, Publication No. F2010/000719-1. Adelaide.

Zampatti, B. P. and Leigh, S. J. (2013a) Effects of flooding on recruitment and abundance of Golden Perch (*Macquaria ambigua ambigua*) in the lower River Murray. *Ecological Management and Restoration* **14**, 135-143.

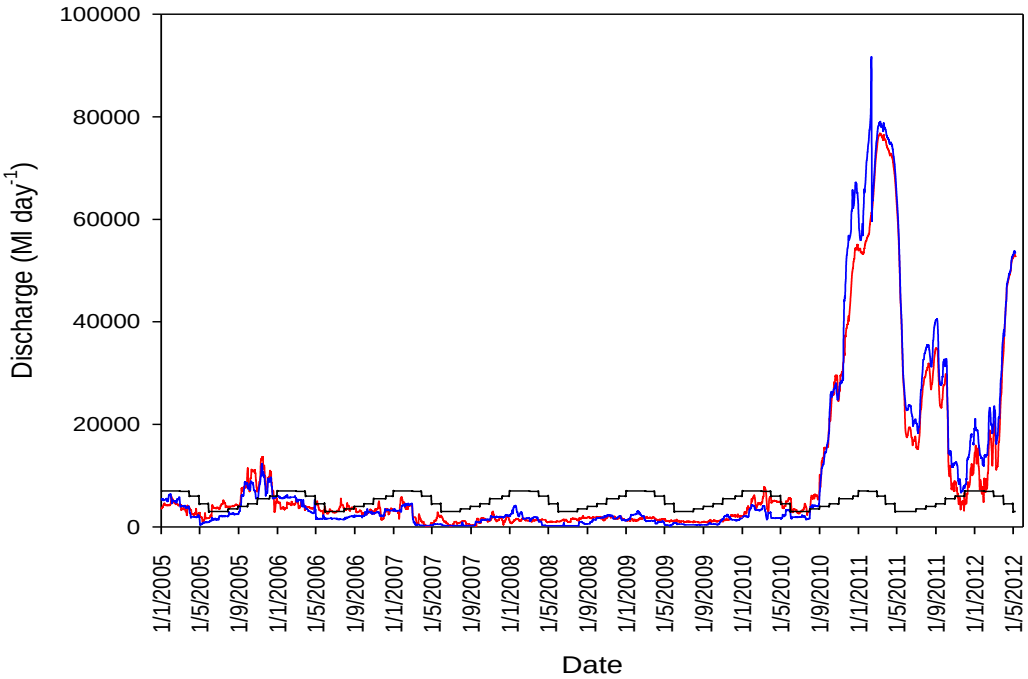
Zampatti, B.P. and Leigh, S.J. (2013b) Within-channel flows promote spawning and recruitment of golden perch, *Macquaria ambigua ambigua*: implications for environmental flow management in the River Murray, Australia. *Marine and Freshwater Research* **64(7)**, 618-630.

Zeug, S.C. and Winemiller, K.O. (2008) Relationships between hydrology, spatial heterogeneity, and fish recruitment dynamics in a temperate floodplain river. *River Research and Applications* **24(1)**, 90-102.

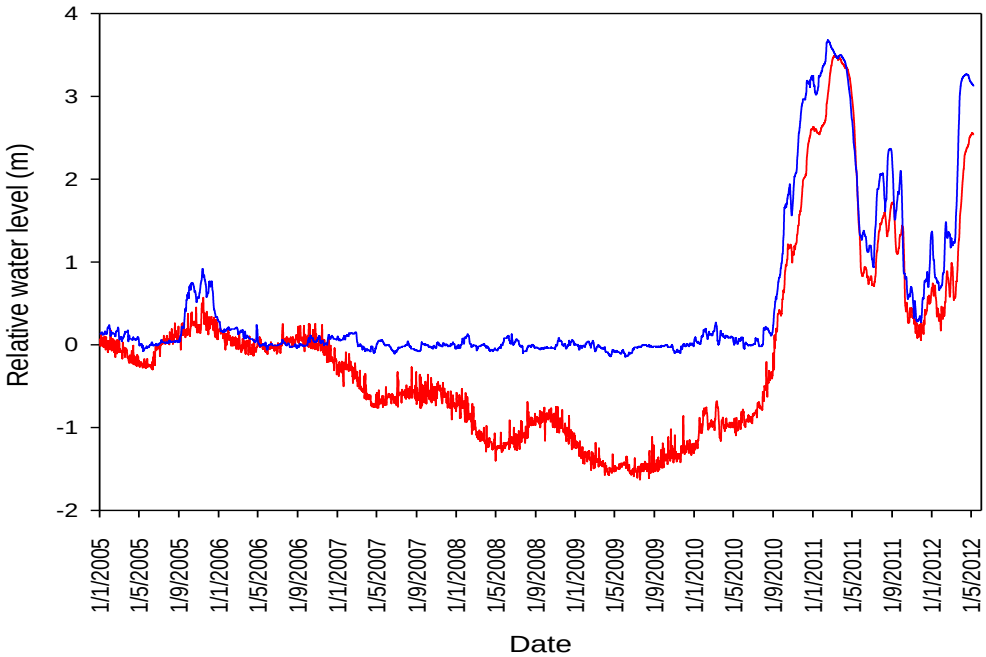
APPENDICES



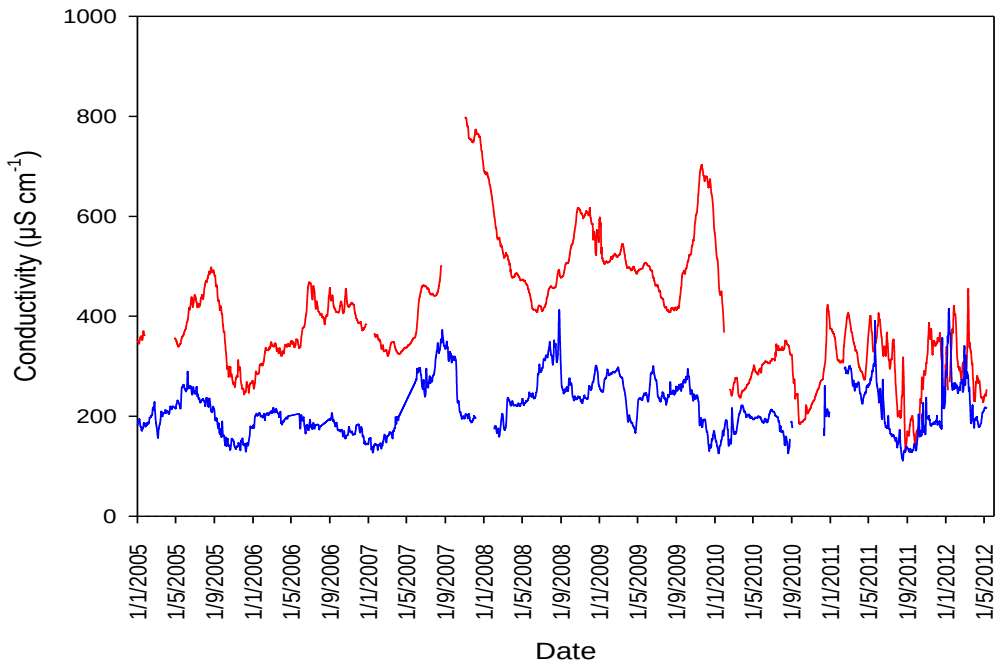
Appendix I. Map showing the location of the Murray–Darling Basin and the major rivers that comprise the southern Murray-Darling Basin, showing the numbered Locks and Weirs (up to Lock 26, Torrumbarry), the Darling, Lachlan, Murrumbidgee, Edward–Wakool, Campaspe and Goulburn rivers and Lake Victoria,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



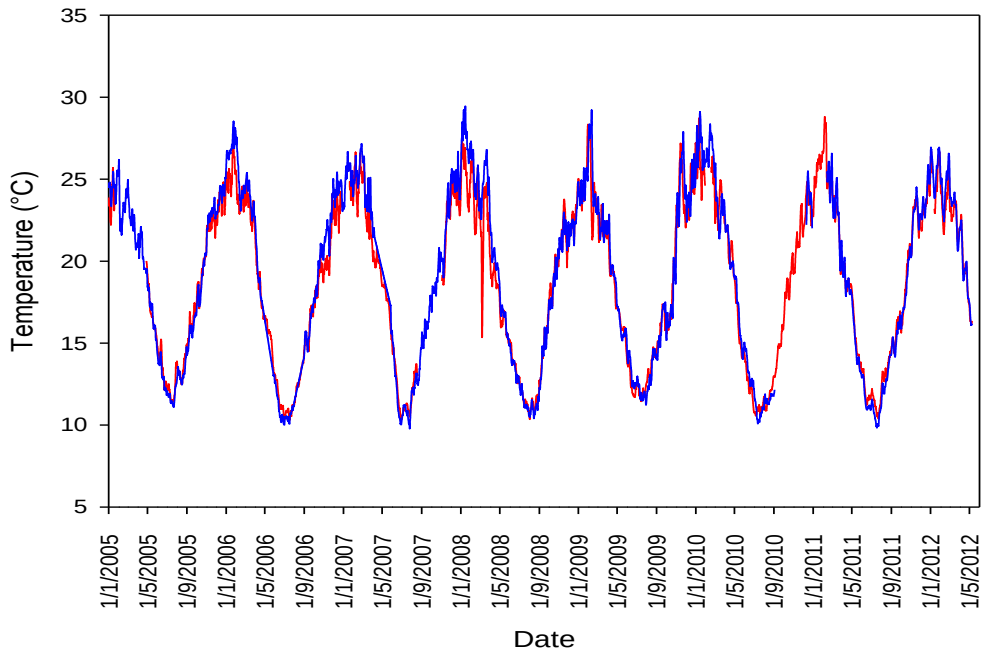
Appendix II. Comparison of the mean daily discharge (ML d^{-1}) from 2005-2012 at Sites 1 (red line) and 2 (blue line) in the lower River Murray. The seasonal entitlement allocation is presented in black line.



Appendix III. Comparison of the mean daily change in water level (mAHD, AHD = Level relative to Australian Height Datum) from 'normal' pool height (Site 1: 0.75 m, Site 2: 16.3 m) from 2005 to 2012, at Sites 1 (red line) and 2 (blue line) in the lower River Murray.



Appendix II. Comparison of the mean daily conductivity (µS cm^{-1} at 25 °C) from 2005 to 2012, at Sites 1 (red line) and 2 (blue line) in the lower River Murray.



Appendix III. Comparison of mean daily water temperature from 2005 to 2012, at Sites 1 (red line) and Site 2 (blue line) in the lower River Murray.

Appendix IVI. Pairwise comparison of larval fish assemblages in the lower River Murray among years for each site. Significant difference (p < 0.0033) in bold

	Site 1		Site 2	
	t	*p* (perm)	t	*p* (perm)
2005-06 vs 2006-07	1.962	0.016	2.056	**0.003**
2005-06 vs 2007-08	2.920	**0.001**	3.432	**0.001**
2005-06 vs 2008-09	4.087	**0.001**	4.060	**0.001**
2005-06 vs 2010-11	6.758	**0.001**	5.277	**0.001**
2005-06 vs 2011-12	4.872	**0.001**	3.746	**0.001**
2006-07 vs 2007-08	1.777	0.033	2.106	0.005
2006-07vs 2008-09	2.766	**0.001**	3.233	**0.001**
2006-07 vs 2010-11	6.100	**0.001**	6.073	**0.001**
2006-07 vs 2011-12	4.110	**0.001**	3.784	**0.001**
2007-08 vs 2008-09	2.484	**0.001**	3.231	**0.001**
2007-08 vs 2010-11	5.863	**0.001**	6.581	**0.001**
2007-08 vs 2011-12	4.473	**0.001**	5.179	**0.001**
2008-08 vs 2010-11	6.197	**0.001**	6.275	**0.001**
2008-09 vs 2011-12	4.985	**0.001**	4.874	**0.001**
2010-11 vs 2011-12	3.421	**0.001**	3.672	**0.001**

Appendix VI. SIMPER analysis for larval fish assemblage comparison between 2011 and other years for a) Site1 and b) Site 2 in the lower River Murray. Results are based on fourth root transformed data. Mean abundance is the relative abundance corrected by flow (mean per trip). CR (consistency ratio) indicates the consistency of differences in abundance between years (shown by PERMANOVA) attributable to individual species. A cumulative cut-off of 75% was applied. Mean dissimilarity is expressed as a percentage ranging between 0% (identical) and 100% (totally dissimilar).

a)

	Mean Abundance				Mean dissimilarity = 56.23
Species name	**2005**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Australian smelt	4.14	1.50	1.37	25.26	25.26
Carp gudgeon	4.25	1.55	1.32	23.58	48.84
Bony herring	3.63	1.95	1.21	21.75	70.59
Flathead gudgeon	2.59	2.39	0.82	12.72	83.31
					Mean dissimilarity = 56.95
Species name	**2006**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Australian melt	3.86	1.50	1.04	27.96	27.96
Bony herring	3.09	1.95	1.17	21.33	49.29
Carp gudgeon	3.10	1.55	1.18	18.99	68.27
Flathead gudgeon	4.13	2.39	1.19	18.11	86.38
					Mean dissimilarity = 60.00
Species name	**2007**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Australian smelt	4.31	1.50	1.16	26.56	26.56
Flathead gudgeon	4.53	2.39	1.40	21.83	48.39
Carp gudgeon	3.71	1.55	1.26	20.42	68.81
Bony herring	1.34	1.95	1.25	13.96	82.77
					Mean dissimilarity = 54.90
Species name	**2008**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Flathead gudgeon	5.67	2.39	1.59	25.61	25.61
Carp gudgeon	3.79	1.55	1.24	19.69	45.30
Australian smelt	2.59	1.50	1.18	16.94	62.24
Bony herring	2.25	1.95	1.32	15.58	77.82
					Mean dissimilarity = 72.32
Species name	**2010**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Flatheadgudgeon	2.30	2.39	0.81	18.12	18.12
Golden perch	1.48	1.69	0.92	15.58	33.70
Carp gudgeon	1.53	1.55	0.88	14.88	48.58
Bony herring	0.31	1.95	0.85	14.44	63.02
Australian smelt	0.26	1.50	0.87	9.95	72.97
Free embryos	1.57	0.00	0.71	9.33	82.31

b)

	Mean Abundance				Mean dissimilarity = 63.93
Species name	**2005**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Carp gudgeon	3.27	1.24	1.27	22.67	22.67
Australian smelt	2.57	0.97	1.18	18.52	41.19
Flathead gudgeon	2.89	1.99	0.99	17.75	58.93
Bony herring	1.27	1.50	0.99	14.41	73.35
Golden perch	0.52	1.33	0.90	11.18	84.53
					Mean dissimilarity = 62.86
Species name	**2006**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Australian smelt	3.13	0.97	1.08	25.61	25.61
Bony herring	2.07	1.50	1.03	18.29	43.90
Carp gudgeon	2.51	1.24	1.18	18.28	62.18
Flathead gudgeon	2.90	1.99	1.07	16.74	78.92
					Mean dissimilarity = 64.72
Species name	**2007**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Australian smelt	4.89	0.97	1.41	30.24	30.24
Bony herring	3.52	1.50	1.25	20.13	50.36
Carp gudgeon	3.60	1.24	1.36	19.09	69.45
Flathead gudgeon	3.81	1.99	1.21	16.75	86.21
					Mean dissimilarity = 67.35
Species name	**2008**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Australian smelt	4.63	0.97	1.30	39.85	39.85
Flathead gudgeon	2.11	1.99	1.07	16.17	56.02
Bony herring	1.71	1.50	1.07	15.51	71.54
Carp gudgeon	1.43	1.24	1.08	13.27	84.81
					Mean dissimilarity = 79.21
Species name	**2010**	**2011**	**CR**	**Contribution (%)**	**Cumulative contribution (%)**
Carp	2.32	0.12	0.87	19.90	19.90
Flathead gudgeon	1.54	1.99	0.91	17.82	37.72
Golden perch	1.86	1.33	0.92	16.31	54.03
Bony herring	0.95	1.50	0.89	13.27	67.30
Carp gudgeon	0.77	1.24	0.79	11.09	78.40

Appendix VII. DistLM sequential results indicating which environmental variable significantly contributed most to the relationship with the multivariate larval fish assemblage data cloud (biological data). Proportion of the variation explained (Prop) and cumulative variation explained (Cumul).

Variable	R^2 res.df	Pseudo-F	P	Prop.	Cumul.
Water discharge	0.190	18.32	**0.001**	0.190	0.190
Temperature	0.277	9.316	**0.001**	0.087	0.278
Relative water level	0.338	6.938	**0.001**	0.060	0.338
Conductivity	0.366	3.320	**0.005**	0.028	0.366