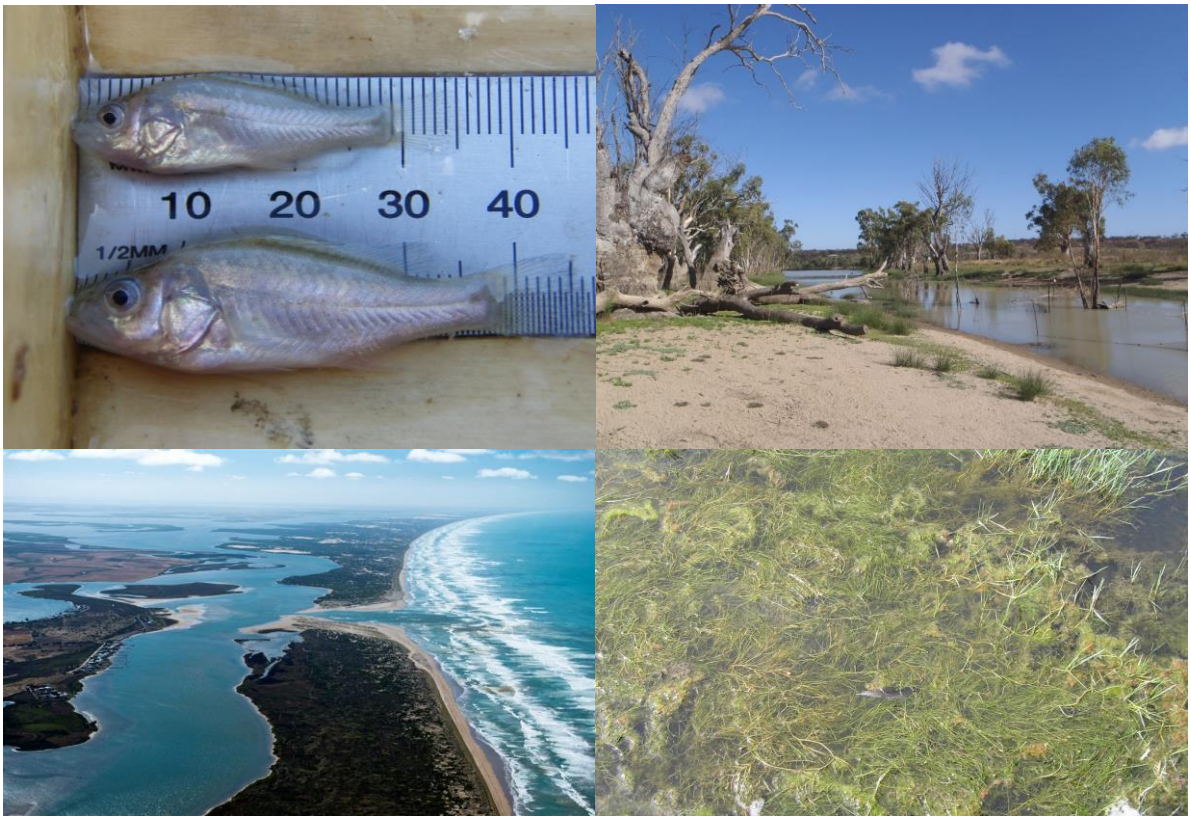
Australian Government
Commonwealth Environmental Water Office

Monitoring ecological response to Commonwealth environmental water delivered to the Lower Murray River in 2013-14

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



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Cover photos: fish, wetland and *Ruppia tuberosa* (SARDI); Murray Mouth (DEWNR).

Disclaimer

This report should be attributed as 'Monitoring Ecological Response to Commonwealth Environmental Water Delivered to the Lower Murray River in 2013-14. Final report prepared for the Commonwealth Environmental Water Office.'

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ACKNOWLEDGEMENTS

This study was funded by the Commonwealth Environmental Water Office (CEWO) in 2013-14 with in kind contributions from the South Australian Research and Development Institute (SARDI), Department of Environment, Water and Natural Resources (DEWNR), and Environment Protection Authority South Australia (EPA).

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EXECUTIVE SUMMARY

This project focuses on the intervention monitoring of the ecological responses to Commonwealth environmental water delivered to the Lower Murray River in 2013-14. The Lower Murray River encompasses a wide range of aquatic habitats that support diverse species of native flora and fauna. The ecosystems in the Lower Murray are 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 Murray River suffered severe stress. The 2010 flood and subsequent high flows in 2011 and 2012 have led to some ecosystem improvement.

In 2013-14, ~480 GL of Commonwealth environmental water was delivered to the Lower Murray River, Lower Lakes and Coorong, in conjunction with other environmental flows (e.g. flows through the Murray–Darling Basin Authority *The Living Murray* Initiative). The flow releases to South Australia were coordinated through a series of watering events across the southern connected Basin to achieve multi-site environmental outcomes. Following the decline of unregulated flows, Commonwealth environmental water delivery throughout November (3,300–4,500 ML day^{-1}) helped to maintain river flow at ~13,000 ML day^{-1} and the delivery of ~5,300–7,300 ML day^{-1} in late December created a flow pulse of ~14,300 ML day^{-1} in the Lower Murray River. Commonwealth environmental water also supplemented freshwater flows to the Lower Lakes and barrage releases to the Coorong.

The current project investigated key ecological responses during 2013-14 in the main channel, wetlands, Lower Lakes and the Coorong, in line with expected ecological outcomes of Commonwealth environmental watering in the Lower Murray River. These included:

- golden perch (*Macquaria ambigua ambigua*) spawning and recruitment;
- fish lateral movement;
- dissolved and particulate material transport; and
- Coorong modelling for *Ruppia tuberosa* and fish habitat.

For each component, monitoring/modelling was conducted to address questions and test hypotheses based on our conceptual understanding of the life history of

relevant biota and ecological processes, and the responses that might be expected from the flow scenarios and environmental water delivery in 2013-14. This report provides a summary of the above studies and a synthesis of the ecological outcomes of environmental watering in the Lower Murray River in 2013-14.

Key ecological outcomes

Monitoring in 2013-14 identified a number of ecological responses associated with the delivery of Commonwealth environmental water in the Lower Murray River. Key outcomes are summarised in Table 1.

Table 1. Summary of CEWO expected short-term (one-year) ecological outcomes, key results, and evaluation questions and answers (blue text) for environmental water releases to the Lower Murray River during 2013-14. CEWO – Commonwealth Environmental Water Office; CEW – Commonwealth environmental water; TLM – The Living Murray.

Indicator	CEWO expected outcomes	Monitoring/modelling results	Evaluation question
Golden perch spawning and recruitment	Enhanced fish reproduction; Increased biotic dispersal	In 2013-14, the delivery of CEW contributed to elevated flows (i.e. above entitlement) in the Lower Murray River in late spring and throughout summer. Golden perch spawned in the lower Murray over this period, predominantly on the descending limb of a within-channel flow pulse, and low abundances of fish were recruited through to young-of-year (YOY) in the floodplain and gorge geomorphic regions of the Lower Murray River.	Did CEW delivery in 2013-14 support the reproduction (spawning and recruitment) of flow-cued spawning native fish species (i.e. golden perch) in the Lower Murray River? CEW contributed to a flow regime that supported golden perch spawning and recruitment to YOY (age 0+).
Fish lateral movements between the main channel and connected wetlands	Increased hydrological connectivity; Increased biotic dispersal	In 2013-14, a diverse fish assemblage was recorded moving between the main river channel and wetlands. Patterns of lateral fish movement were complex and highly variable among flow phases, and changes were not consistent among wetlands. Movements of most species at each wetland during each flow phase were generally bidirectional and demonstrated no clear, consistent pattern relative to changes in flow conditions. Fish assemblages moving in and out of wetlands were driven primarily by inlet flow, flow in the main channel and water temperature.	Did CEW delivery in 2013-14 increase the area of inundated aquatic habitat and enhanced the movements of native fish species? CEW did not have noticeable effects on inlet flow and inlet height of the two wetlands. Increased native fish lateral movement was observed during CEW delivery for Hart Lagoon; however, movement was generally bidirectional and demonstrated no clear pattern relative to changes in flow conditions.
Transport of salt, nutrients, other dissolved and particulate	Increased salt transport and export;	The modelling outputs suggest that during 2013-14, environmental watering resulted in significant increases in the transport and export of salt, nutrients, chlorophyll	Did CEW delivery in 2013-14 contribute to the transport of salt, nutrients and other dissolved and particulate matter through the Murray Mouth?

Indicator	CEWO expected outcomes	Monitoring/modelling results	Evaluation question
material through the Murray River (Lock 1 to southern ocean)	Increased sediment/nutrient transport; Providing end of system flows	(phytoplankton) and suspended solids through the Lower Murray River, Lower Lakes and Murray Mouth. A majority of the total exports were achieved during an unregulated flow event (July to October) prior to the period of environmental water delivery. However, thereafter the additional exports were almost entirely associated with environmental water deliveries and overall environmental water contributed up to 45% of total exports of dissolved and particulate matter. It was evident that both CEW and TLM environmental water contributed to the transport and export of dissolved and particulate matter, although a majority was associated with CEW due to greater volumes of water provided in 2013-14.	CEW resulted in significant increases in the transport and export of salt, nutrients, chlorophyll (phytoplankton) and suspended solids through the Lower Murray River, Lower Lakes and Murray Mouth.
Coorong modelling to evaluate environmental water benefits to *Ruppia tuberosa* and fish habitat	Increased hydrological connectivity; Reduced salinity; Providing end of system flows; Increased within ecosystem diversity; Increased resistance	*R. tuberosa*: Environmental watering in 2013-14 resulted in a significant increase in the chance of propagule bank replenishment for the North Lagoon of the Coorong compared to the model outputs without environmental water. However, environmental water delivery made little difference to the chance of propagule bank replenishment in the South Lagoon. Fish: Habitat suitability generally improved for all fish species investigated. Habitat suitability for the least salt tolerant species, mullloway, increased by 25% and habitat extended southward up to 40 km due to the deliveries of CEW and TLM water.	Did CEW delivery in 2013-14 provide benefits for *R. tuberosa* and fish habitat in the Coorong? CEW resulted in a significant increase in the chance of propagule bank replenishment for the North Lagoon of the Coorong and improved habitat suitability for fish.

Key learning and management implications

Based on the outcomes from 2013-14 monitoring, and current knowledge of nutrient dynamics and flow related biology/ecology in the Lower Murray River, Lakes and Coorong, the following points should be considered with regard to environmental water planning and flow management.

Lower Murray River

Spawning of golden perch in the Lower Murray River is associated with variability in flows (contained within-channel or overbank) in mid-late spring and throughout summer, generally at water temperatures >20°C. A pulse of water with the appropriate timing and adequate magnitude and duration may enhance and extend the presence of golden perch larvae which may in turn lead to enhanced recruitment in the Lower Murray River. The source and continuity of the delivered water may also play an important role in the outcomes achieved.

Environmental water delivery in 2013-14 did not have any noticeable influence on the lateral movement of fish assemblages, suggesting that the magnitude of water delivery (up to ~14,000 ML day^{-1}) may be insufficient for enhancing fish movements between the main channel and connected wetlands in the Lower Murray River. Future environmental water deliveries of greater magnitude that influence wetland inlet flow and height are expected to support and enhance lateral fish movement in the Lower Murray River. These flows could be delivered in conjunction with high unregulated flow events to maximise variability in wetland inlet flows.

Environmental flow delivery to the Lower Murray, Lower Lakes and Murray Mouth during low to moderate flow periods (e.g. 10,000–40,000 ML day^{-1}) will increase the transport of dissolved and particulate matter through the system and significantly reduce salinity concentrations within the Murray Mouth region. Deliveries during extended low flow periods are likely to have greater impacts on concentrations of dissolved and particulate matter than periods with antecedent moderate flow conditions.

Coorong

The volumes of environmental water currently available (~480 GL year^{-1}) will have limited benefit to *Ruppia tuberosa* populations in the South Lagoon unless delivered in conjunction with an unregulated flow event. Even in this situation the unregulated flow will need to be of sufficient duration to provide barrage outflows during November and early December and environmental water used to manage flow recession to reduce the rate of water level decline in the South Lagoon to reduce the risk of stranding. For environmental flows alone to have a significant benefit, much larger volumes of water will be required than are currently available.

Freshwater inflow is pivotal for maintaining estuarine fish habitat and populations in the Coorong. Environmental watering could be managed to maintain the connectivity and extend barrage outflows to improve the quality and extent of fish habitat in the Coorong. Environmental watering during the summer months or in years with less barrage flows and higher salinity will provide a much larger effect on habitat improvement. While maintaining barrage releases and connectivity is important throughout the year, flow delivery is critical during late spring/summer as this period corresponds to the spawning and recruitment season of most estuarine fish species in the Coorong. Environmental flows could potentially help to maintain a favourable salinity gradient, enhancing productivity and improving connectivity to facilitate fish recruitment.

More specific recommendations for future research and monitoring for the Lower Murray River and Coorong are provided in Section 7.

1 INTRODUCTION

1.1 General background

River regulation and flow modification have severely impacted riverine ecosystems throughout the world (Kingsford 2000; Bunn and Arthington 2002; Tockner and Stanford 2002). Natural flow regimes play a critical role in maintaining ecological integrity of floodplain rivers (Junk *et al.* 1989; Poff *et al.* 1997; Puckridge *et al.* 1998; Lytle and Poff 2004). Therefore, ecological restoration of river systems often involves environmental water allocations to re-establish key components of the natural flow regime in order to restore important ecological processes (Poff *et al.* 1997; Arthington *et al.* 2006). Understanding biological and ecological responses to flow provides critical knowledge to underpin environmental flow management to achieve the best ecological outcomes (Walker *et al.* 1995; Arthington *et al.* 2006).

The Lower Murray River represents a significant ecological asset to be targeted for environmental watering (DEWNR 2013). The complex system includes the main river channel, anabranches, floodplain/wetlands, billabongs, stream tributaries and the Lower Lakes, Coorong and Murray Mouth, which provide a range of water dependent habitats and support significant flora and fauna. The distribution and abundance of all aquatic biota is influenced by the flow regime which plays an overarching role in driving riverine ecosystem structure and function (Poff and Allan 1995; Sparks *et al.* 1998). During the decadal drought in the Murray–Darling Basin (MDB) (2001–2010), the ecosystem of the Lower Murray River was under severe stress; much of the biota declined and ecosystem resilience was compromised (e.g. Noell *et al.* 2009; Nicol 2010; Zampatti *et al.* 2010). A natural flood in late 2010 and the following year's high flows with environmental water deliveries have contributed to ecological recovery (e.g. Gehrig *et al.* 2012; Nicol *et al.* 2013; Ye *et al.* 2013b; 2015a; 2015b).

Since 2011-12, significant volumes of Commonwealth environmental water have been delivered to the Lower Murray River, Lower Lakes and Coorong, in conjunction with other environmental flows (e.g. flows through the Murray–Darling Basin Authority (MDBA) *The Living Murray* (TLM) Initiative), to facilitate post drought ecosystem recovery and restore ecological health (www.environment.gov.au/ewater/). The flow releases to South Australia were coordinated through a series of watering events

across the southern connected Basin to achieve multi-site environmental outcomes (www.environment.gov.au/ewater/).

1.2 Hydrology and environmental watering in 2013-14

The MDB is a highly regulated river system, particularly in the southern Basin, where the natural flow regimes have been substantially modified, leading to decreased hydrological variability, increased water level stability and reduced floodplain inundation (Maheshwari *et al.* 1995; Richter *et al.* 1996). The Lower Murray River is heavily modified by upstream diversions and extraction, and a series of 11 low-level (<3 m) weirs constructed in the 1930s–1940s, which changed a connected flowing river to a series of weir pools (Walker 2006). These have had a profound impact on riverine processes and the ecology of the Lower Murray River (Walker 1985; Walker and Thoms 1993).

From 1996 to 2010, the MDB experienced a severe drought during which inflows into the Murray River system were approximately 40% of the historical mean (MDBA 2011). The drought was broken in late 2010 by a significant overbank flow, reaching a peak of approximately 93,000 ML day^{-1} in February 2011 in the Lower Murray River. In the subsequent two years, flow remained high although largely confined within the river channel (< 50,000 ML day^{-1}) (Figure 1).

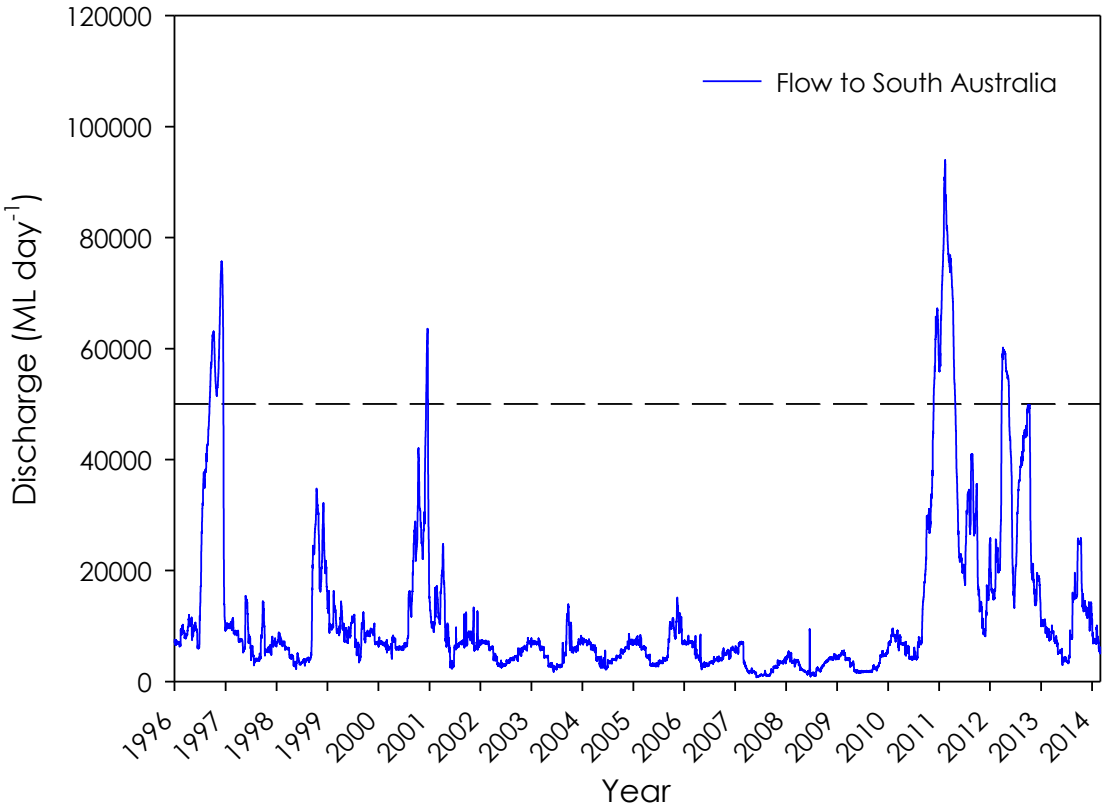


Figure 1. Daily flow (ML day^{-1}) in the Lower Murray River at the South Australian border from January 1996 to April 2014. Dotted line represents approximate bankfull flow in the main channel of the Lower Murray River.

During 2013-14, ~480 GL of Commonwealth environmental water was delivered to the Lower Murray River, Lower Lakes and Coorong, in conjunction with other environmental flows (e.g. flows through the MDBA TLM initiative). Following the decline of unregulated flows, Commonwealth environmental water delivery throughout November (3,300–4,500 ML day^{-1}) helped to maintain river flow at ~13,000 ML day^{-1} and the delivery of ~5,300–7,300 ML day^{-1} in late December created a flow pulse of ~14,300 ML day^{-1} in the Lower Murray River (Figure 2). TLM environmental water was also delivered during these periods, particularly in November. Details on the physical sources of environmental water (e.g. Lake Victoria, mid-Murray River or Darling River) remained unclear. Commonwealth environmental water also supplemented freshwater flows to the Lower Lakes and barrage releases to the Coorong.

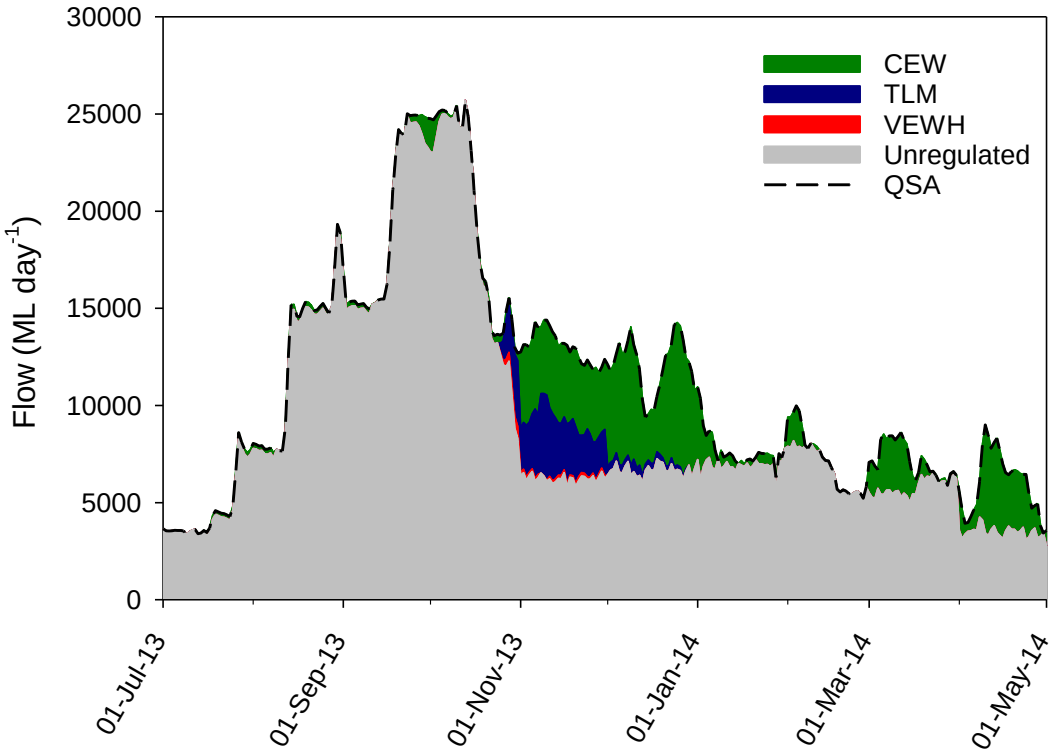


Figure 2. Murray River flow to South Australia from July 2013 to May 2014. Environmental water includes Commonwealth environmental water (CEW), The Living Murray (TLM) and Victorian Environmental Water Holder (VEWH).

1.3 CEWO expected ecological outcomes, monitoring tasks and evaluation questions

Intervention monitoring was conducted to evaluate the ecological outcomes of Commonwealth environmental water delivery to the Lower Murray River in 2013-14. Four indicators were selected, in line with the expected short-term (one year) ecological outcomes, including 1) golden perch (Macquaria ambigua ambigua) spawning and recruitment; 2) lateral movements of fish; 3) dissolved and particulate material transport; and 4) Coorong modelling for *Ruppia tuberosa* and fish habitat. Through four respective tasks, targeted investigations were carried out to explore flow related ecological responses in the Lower Murray River (i.e. the main channel and associated floodplain/wetland; Coorong). We proposed to test a series of hypotheses based on our conceptual understanding of the life histories of relevant biota and ecological processes, and the expected responses from the flow scenarios with environmental water delivery in 2013-14. The following conceptual diagram illustrates our current understanding of how river ecosystems are affected by the flow

regime, subject to flow management and climate effects, and how the proposed complementary monitoring components (tasks) contribute toward a holistic understanding of ecosystem responses to flow management and ecological benefits (Figure 3). More details are presented below, for each indicator (monitoring task), regarding the relevant CEWO expected short-term ecological outcomes, hypotheses and evaluation questions.

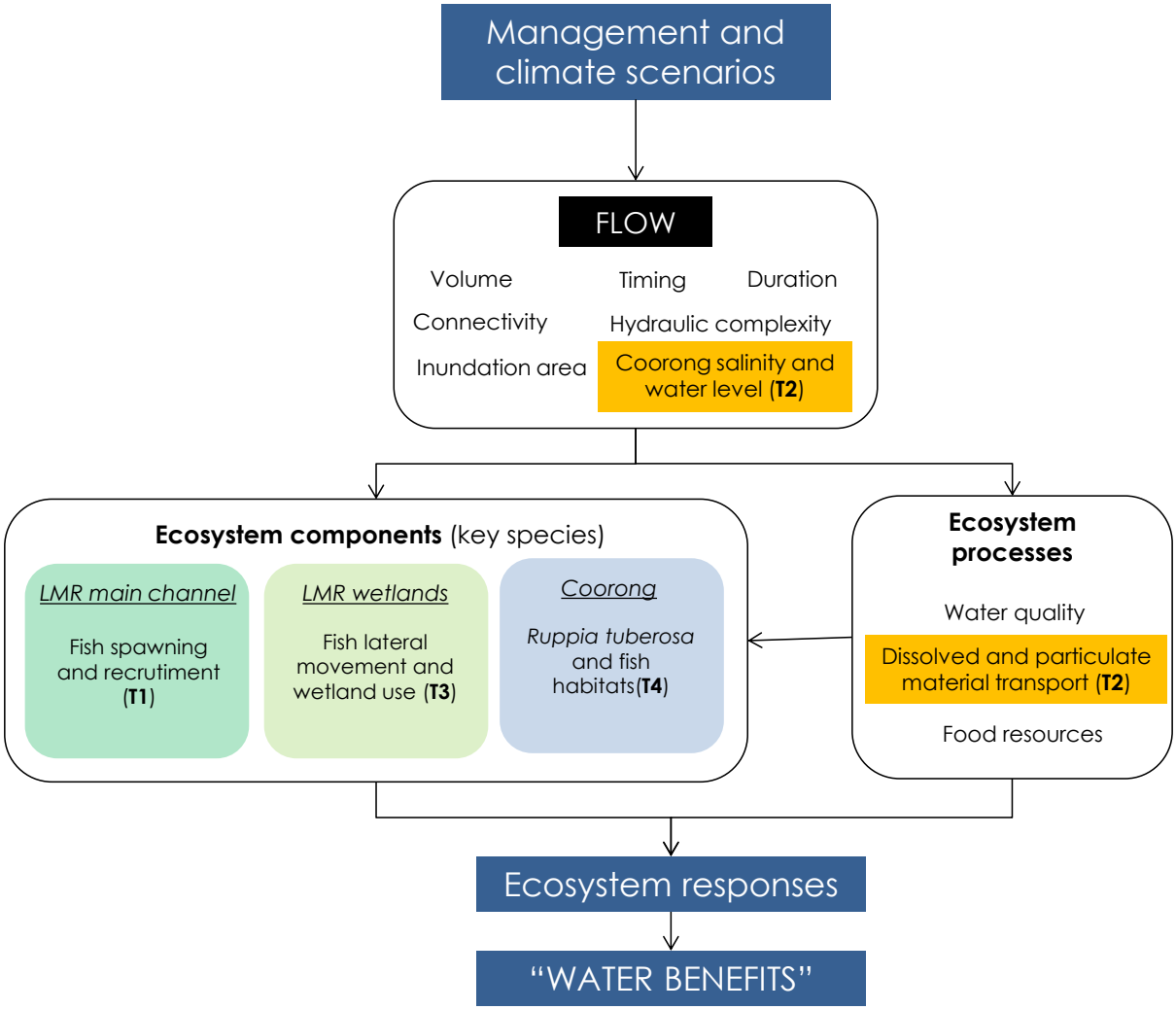


Figure 3. Conceptual diagram of how river ecosystems are affected by the key ecosystem driver (flow regime), subject to flow management and climate effects, and how complementary monitoring components (tasks) contribute toward a holistic understanding of ecosystem responses to flow management and ecological benefits in the Lower Murray River (LMR) (Note tasks, T1–T4, are within ecosystem components and in highlighted orange boxes under flow and ecosystem processes).

Golden perch spawning and recruitment (Task 1)

CEWO expected outcome:

- enhanced fish reproduction;
- increased biotic dispersal.

Hypotheses: Increased flow in the Lower Murray River (i.e. >15,000 ML day^{-1}) in spring/summer will promote spawning and lead to the presence of larvae of golden perch, a flow-cued spawner.

Increased flow in the Lower Murray River (i.e. >15,000 ML day^{-1}) in spring/summer will result in recruitment of young-of-year (YOY) golden perch in the Lower Murray River.

The origin of YOY and larvae spawned during increased flows will be influenced by the source of flow, i.e. the Darling River, Murrumbidgee River, Murray River upstream of the Darling Junction or tributaries of the mid Murray (e.g. Goulburn River).

Evaluation question: Has environmental watering supported the reproduction (i.e. spawning and recruitment) of native fish in the Lower Murray River that are flow-cued spawners?

Lateral movements of fish (Task 3)

CEWO expected outcomes:

- increased hydrological connectivity;
- increased biotic dispersal.

Hypotheses: Sustained flows will maintain the lateral movement of fish (from the River channel to the connected wetlands), particularly small bodied species.

Extended high flow conditions as a result of environmental watering will increase inundated aquatic habitat, potentially providing a nursery ground for fish.

Changes in environmental conditions, i.e. water level, water temperature, salinity, flow direction, flow velocity) associated with different phases of environmental water delivery to South Australia will influence the abundance and diversity of native and

exotic fish species moving between the main channel and off-channel wetland habitats in the Lower Murray River.

Evaluation question: Has environmental watering increased the area of inundated aquatic habitat and enhanced the movements of native fish between the river channel and connected wetlands?

Dissolved and particulate material transport (Task 2)

CEWO expected outcomes:

- increased salt transport and export;
- increased sediment/nutrient transport;
- providing end of system flows.

Hypotheses:

Salt transport: environmental watering will increase the mobilisation of salts from the Basin and increase the transport of salt passing from Lock 1 through the Lower Murray River, Lower Lakes and Murray Mouth.

Nutrient transport: environmental watering will increase the mobilisation of nutrients from the Basin and increase nutrient loads passing from Lock 1 through the Lower Murray River, Lower Lakes and Murray Mouth.

Particulate organic matter: environmental watering will increase the mobilisation and production of particulate organic matter and increase particulate organic matter loads passing from Lock 1 through the Lower Murray River, Lower Lakes and Murray Mouth.

Particulate inorganic matter: environmental watering will increase the re-suspension of inorganic matter from the river bed and thus increase the transport of particulate inorganic matter from Lock 1 through the Lower Murray River, Lower Lakes and Murray Mouth.

Evaluation question: Has environmental watering contributed to the transport of salt, nutrients and other dissolved and particulate matter through the Murray Mouth?

Coorong modelling (Ruppia tuberosa and fish habitat) (Task 4)

CEWO expected outcomes:

- increased hydrological connectivity;
- providing end of system flows;
- reduced salinity;
- increased within ecosystem diversity;
- increased resistance.

Hypotheses: Increased freshwater flow through the barrages and into the Coorong due to environmental watering will reduce salinity and increase water levels in the Coorong, thus enhancing *R. tuberosa* and fish habitats.

Evaluation question: Has environmental watering provided benefits for *R. tuberosa* and fish habitat in the Coorong?

This report presents the findings and outcomes of environmental watering in the Lower Murray River (South Australia) during the 2013-14 intervention monitoring. The results from the *R. tuberosa* modelling are reported to the end of 2013 because the growing season of *R. tuberosa* in the Coorong commences in late autumn and ends between mid-spring and early summer.

2 BACKGROUND BIOLOGY AND ECOLOGY

2.1 Golden perch spawning and recruitment

River regulation and alteration of the natural flow regime have contributed to native fish population declines in the MDB (Barrett 2004). River regulation may impact fish directly through loss of spawning cues and barriers to migration and dispersion, and indirectly through effects on fish habitat and food resources. Restoring natural flow regimes with management strategies such as environmental water allocations have become a focus of ecosystem restoration in the MDB (MDBA 2012; Koehn *et al.* 2014). However, for flow restoration to benefit aquatic ecosystems, including fish, requires an empirical understanding of relationships between hydrology, life history and population dynamics (Arthington *et al*. 2006).

Golden perch and silver perch (*Bidyanus bidyanus*) are two species in the southern MDB that are considered to require increased river flow to commence spawning (Humphries *et al.* 1999). Spawning and recruitment of golden perch in the Murray River has been associated with both increases in within-channel flow and overbank flooding (Mallen-Cooper and Stuart 2003; King *et al.* 2009; Zampatti and Leigh 2013a; 2013b). Golden perch are pelagic spawners with buoyant eggs and larvae, which may passively drift considerable distances downstream from where they were spawned.

Understanding the influence of hydrology and the mechanisms that facilitate golden perch reproduction (i.e. spawning and recruitment) is reliant on accurately determining the hydrological conditions at the time and place of crucial life-history processes. For example, to be able to accurately determine the hydrological conditions associated with spawning, the time and place of spawning must be known. This can be achieved by the *in situ* collection of eggs immediately post-spawning and/or by retrospectively determining the spatio-temporal provenance of larval, juvenile and adult fish (i.e. *when* and *where* a fish was spawned).

2.2 Lateral movements of fish

River regulation has a major impact on the ecological processes in riverine systems (Kingsford 2000). Modifications to the magnitude, timing, frequency and duration of freshwater flows have altered hydrological regimes, which has subsequently altered patterns of lateral connectivity between the main channel and off-channel wetlands (Jones and Stuart 2008). Consequently, long established links between environmental cycles and the ecology of wetland flora and fauna has been disrupted and exotic species, such as carp (*Cyprinus carpio*) have benefited (Gehrke *et al.* 1995). Lateral connectivity is important for maintaining native fish populations through increased survival, recruitment, feeding and reproduction opportunities (Junk *et al.* 1989), as off-channel habitats increase habitat availability, complexity and productivity.

A series of low level weirs in the Lower Murray River alter the natural flow and modify the water level within each weir pool. The inundation of many off-channel wetlands in the Lower Murray River is highly dependent on flow and main channel water level. A diverse range of fish species that are known to use off-channel wetlands are, therefore, directly affected by flow discharge and weir pool management. However, knowledge on the lateral movements of freshwater fish between main-channel and wetland habitats in the Murray River is limited.

In 2006, Conallin *et al.* (2011) examined the lateral movement of fish in the Lower Murray River under low-flow conditions. In that study, variations in the lateral movements of adult and juvenile fish were observed but no directional consistency was identified which likely reflected an absence of hydrological cues, due to the lack of flow during the drought. Furthermore, Conallin *et al.* (2011) predicted that directional movements would occur under increased flow conditions. In 2008, under continuing, extreme drought conditions, Conallin *et al.* (2012) found that fish movements into a flow-through wetland were associated with rising water temperature. In contrast, under variable flow conditions in the mid-upper Murray River (north-eastern Victoria), lateral movements of small-bodied native fish were strongly correlated with fluctuations in water level - as water levels rose, fish moved from the main river channel into off-channel habitats, before returning to the main river as water levels subsequently receded (Lyon *et al.* 2010).

Environmental water allocations are increasingly used as a management tool for habitat restoration and natural resource management (King *et al*. 2009). Understanding the influence of different flow scenarios on the lateral movements of fish between main-channel and wetland habitats would facilitate the development of well-informed, flow-related management strategies to (*i*) enhance reproduction and recruitment success of native fish; and (*ii*) inhibit key aspects of the population dynamics of exotic species, such as carp.

The delivery of environmental water to South Australia during summer 2013-14 provided the opportunity to monitor lateral movements of fish in conjunction with managed environmental flows in the Lower Murray River. As such, this task examined the influence of environmental water and associated hydrological stimuli on the lateral movements of native fish between the main-channel of the Lower Murray River and two wetlands in South Australia. The knowledge generated from this monitoring will support adaptive management of environmental flows with the aim of biasing flow management towards benefits for native fish. The specific objectives addressed in this report were to describe the components of fish assemblages attempting to access and exit wetlands in the 'Floodplain' region of the Lower Murray River; and determine if these assemblages differ between wetlands and among four phases of flow.

2.3 Dissolved and particulate material transport

Flow affects habitat and provides resources for aquatic organisms by altering the concentrations and transport of dissolved and particulate matter. Here we consider dissolved and particulate matter to include:

- salinity, which is a measure of total dissolved salts and is a key parameter governing the distribution and abundance of aquatic biota. Salinity is strongly influenced by flow through the alteration of groundwater inputs, evapoconcentration and intrusions of seawater (Brookes *et al.* 2009; Aldridge *et al.* 2011; 2012; Mosley *et al.* 2012).
- dissolved inorganic nutrients, which are essential resources for the growth and survival biota and are readily assimilated (Poff *et al.* 1997). Nitrogen, phosphorus and silica are particularly important because they often control the productivity of aquatic ecosystems. Flow results in the mobilisation and

transport of dissolved nutrients through the leaching of nutrients from dried sediments and dead organic matter.

- particulate organic nutrients (phosphorus and nitrogen), which are those nutrients incorporated into the tissue of living and dead organisms. Flow can influence particulate organic nutrient concentrations and transport through a number of mechanisms, including through increased productivity associated with elevated dissolved nutrient concentrations.
- chlorophyll *a*, which is a measure of phytoplankton biomass, with phytoplankton being an important primary producer of riverine ecosystems. Flow can influence chlorophyll *a* concentrations and transport through increased phytoplankton productivity.
- total suspended solids, which is a measure of the total amount of inorganic and organic particulate matter in the water column. It has a strong influence on light availability, which is important for structuring aquatic ecosystems (Geddes 1984a; 1984b). It is influenced by flow through increased productivity, as well as the mobilisation of inorganic matter from the floodplain and river channel (i.e. resuspension).

Altering the flow regime of riverine systems has had significant consequences for the concentrations and transport of dissolved and particulate matter (Aldridge *et al.* 2012). For example, reduced flow can result in salinisation through evapoconcentration and the intrusion of saline water, reduced sediment transport and increased sedimentation due to deposition, reduced nutrient concentrations due to decreased mobilisation of nutrients from the floodplain, reduced primary productivity because of nutrient limitation; and thus reduced secondary productivity. Such observations have been made in the Murray River, including the Lower Murray River, Lower Lakes and Coorong (Brookes *et al.* 2009; Aldridge *et al.* 2011; 2012; Mosley *et al.* 2012).

Environmental flow deliveries may be used to reinstate some of the natural processes that control the concentrations and transport of dissolved and particulate matter (Aldridge *et al.* 2012). In doing so, these flows may provide ecological benefits through the provision of habitat and resources for biota.

2.4 Coorong modelling

The Coorong is a dynamic estuarine lagoon system located at the terminus of the Murray-Darling Basin in South Australia. It has been heavily impacted by river regulation and water extraction upstream since European settlement; subsequently the current average annual flow has declined by 61% at the Murray Mouth (from 12,333 GL year^{-1} to 4,733 GL year^{-1}; CSIRO 2008). The Coorong has a strong north-south salinity gradient, generally ranging from brackish/marine in the area of the Murray Mouth to hypersaline in the North and South Lagoons (Geddes and Bulter 1984; Geddes 1987). Salinities are spatio-temporally variable and highly dependent on the freshwater inflows from the Murray River, with varied salinities supporting different ecological communities (Brookes *et al.* 2009). In addition, the southern end of the South Lagoon receives small volumes of fresh/brackish water from a network of drains (the Upper South East Drainage Scheme) through Salt Creek.

Freshwater inflow is a crucial driver affecting salinity and water level regimes in the Coorong; these physical-chemical parameters have a strong influence on the habitat and ecological communities including the iconic macrophyte, *Ruppia tuberosa* (Nicol 2005), and a range of fish species in the Coorong (Ye *et al.* 2015c). The hydrodynamic model developed by CSIRO (Webster 2007; 2013) is now available as the *Coorong Hydrodynamic Model* (CHM) v2.1 (Joehnk and Webster 2014) and allows the simulation of salinity and water levels along the Coorong. This was combined with our conceptual understanding of the life-history of *R. tuberosa* and the effects of salinity and water levels on key processes to evaluate the ecological benefit based on flow regimes and 2013-14 environmental watering in the Coorong. In addition, exploratory analysis on the extent of estuarine fish habitat was also undertaken based on the salinity tolerance thresholds of seven key species in the Coorong.

Ruppia tuberosa

Ruppia tuberosa is a submergent halophyte that was historically common in the South Lagoon of the Coorong (Geddes and Brock 1977; Brock 1979; 1981; Paton 1982; 1996; Geddes and Butler 1984; Geddes 1987). It is one of the most salt tolerant angiosperms with a maximum salinity tolerance of 230 g L^{-1} for adult plants (Brock 1982a); however, much lower salinities are required for life cycle completion. Kim *et al.* (2013) reported that salinities lower than 85 g L^{-1} for 15 days are required for germination from seeds

and 125 g L^{-1} for sprouting from turions (a specialised drought resistant asexual propagule produced by aquatic plants). Furthermore, exposure to elevated salinity followed by lower salinity stimulated germination in seeds but reduced viability of turions by over 90% (Kim *et al.* 2013). Brock (1982b) also noted that at elevated salinities *R. tuberosa* did not flower and was restricted to reproducing asexually; therefore, lower salinities are required for the production of seed and subsequent replenishing of the sediment seed bank.

Water levels are also a critical factor for *R. tuberosa* in the South Lagoon of the Coorong. *R. tuberosa* is highly sensitive to desiccation but has high light requirements; therefore, there is a narrow band where the species can occur in the highly turbid South Lagoon (Nicol 2005). *R. tuberosa* colonises areas between 0 and -0.5 metres, level relative to Australian Height Datum (AHD), in May to June in the South Lagoon; areas below -0.5 m AHD are below the euphotic zone and areas above 0 m AHD are prone to desiccation due to wind driven water levels fluctuations (seiching) (Nicol 2005). These water levels need to be maintained until at least mid-November, preferably mid to late December, to ensure the life cycle is completed and the seed bank replenished (Figure 4).

Figure 5 represents the optimal salinity regime for *R. tuberosa* in the South Lagoon. For turions to sprout, salinity needs to be lower than 125 g L^{-1} and 85 g L^{-1} for seed germination for at least 15 days (Kim *et al.* 2013). Salinity needs to be maintained below 100 g L^{-1} for the duration of the growing season to ensure plants reproduce sexually. Whilst seed production is restricted at salinities above 100 g L^{-1}, turions may be produced but understanding the extent to which this occurs requires further study. The maximum salinity thresholds for adult plants (230 g L^{-1}), turion sprouting (125 g L^{-1}), seed germination (85 g L^{-1}), sexual reproduction (100 g L^{-1}) and turion viability (130 g L^{-1}) are also represented.

The information in Figure 4 and Figure 5 are summarised in Figure 6, which provides a conceptual model representing the life-history of *R. tuberosa* in the South Lagoon. The life-history of *R. tuberosa* is represented by five stages; the sediment propagule bank, seedlings, juveniles, asexual adults and sexual adults. The sediment propagule bank consists of seeds and turions. Turions will sprout (in May to June) when inundated with water that has salinity lower than 125 g L^{-1} and seeds will germinate when the salinity is below 85 g L^{-1} (Kim *et al.* 2013). Seedlings will persist and become juveniles providing

the water level is maintained above +0.2 m AHD and the salinity remains below 100 g L^{-1} and in turn juveniles will become asexual adults if the aforementioned conditions are maintained until October. If the salinity remains below 100 g L^{-1} and the water level above +0.2 m AHD until mid-November (preferably mid to late December) the plants will reach sexual maturity and replenish the propagule bank. However, if the salinity exceeds 100 g L^{-1} the plants will not flower but turions may be produced that will partially replenish the propagule bank. If water levels fall below +0.2 m AHD before mid-November plants will die and the propagule bank will not be replenished.

The ecological bottlenecks identified were seed germination and turion sprouting occurring because, during periods of extended barrage closure, the salinity in May and June often did not fall below the thresholds for germination or sprouting (Figure 6). Furthermore, the high salinities experienced over summer during periods of extended barrage closure may have reduced turion viability in the propagule bank. The other bottle necks identified were plants not reaching sexual maturity and the propagule bank not being replenished (Figure 6). Barrage outflows often cease or are reduced during late October to early November, which causes a sudden drop in water level in the South Lagoon, stranding plants that have not reached sexual maturity.

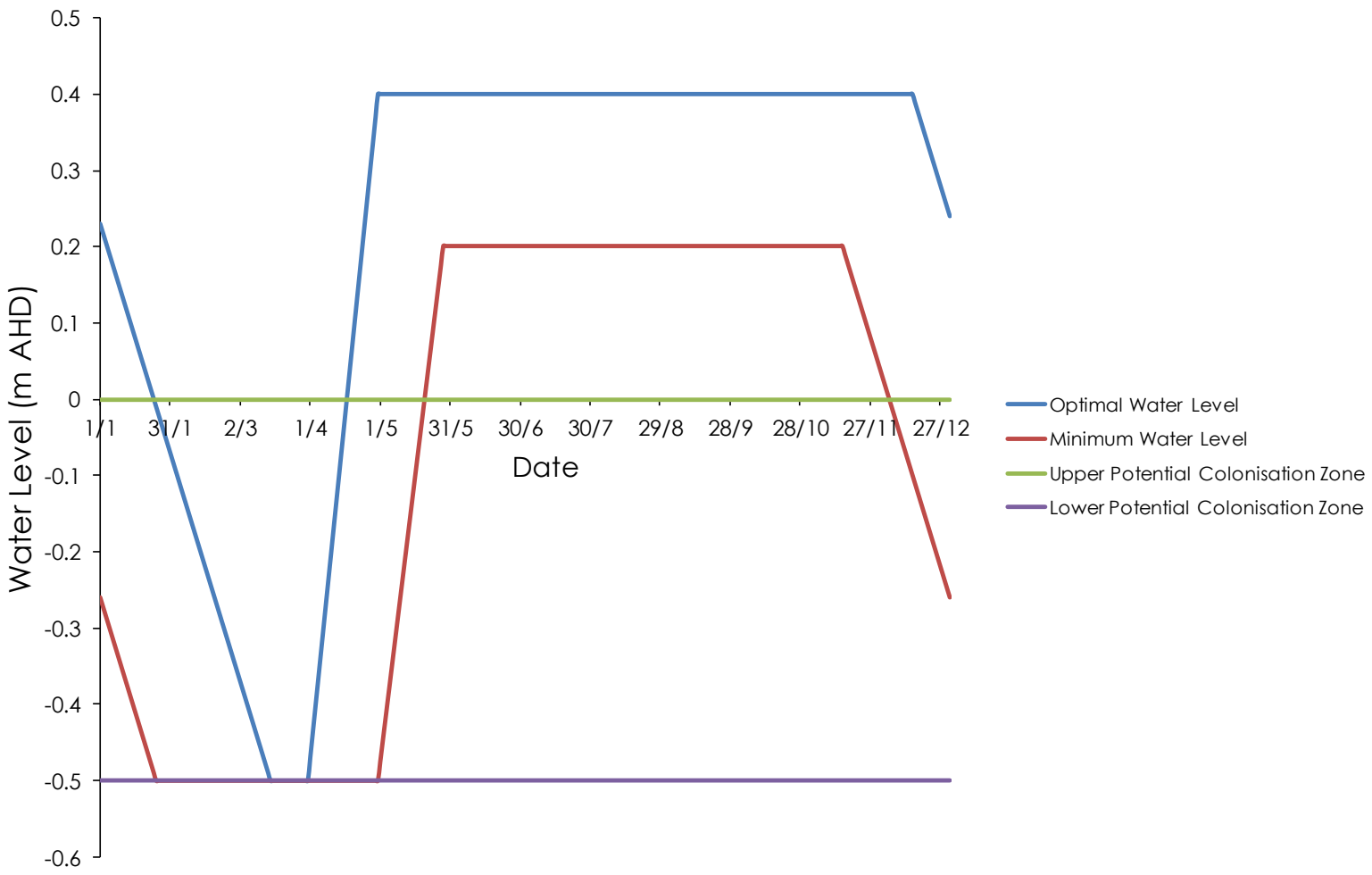


Figure 4. Optimal hydrograph for *Ruppia tuberosa* in the South Lagoon of the Coorong howing the minimum water levels throughout the year and the elevations where colonisation is likely to occur.

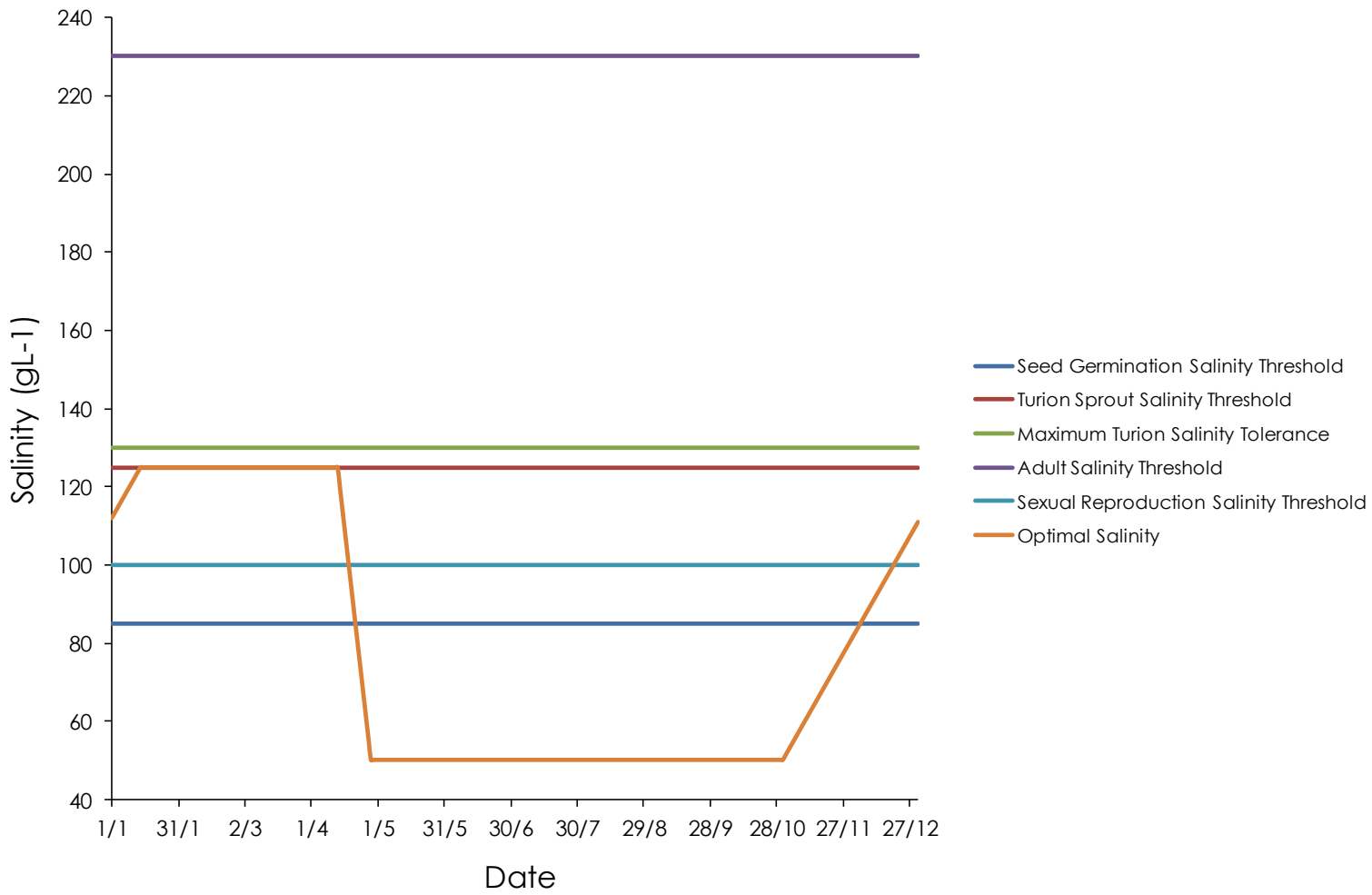


Figure 5. Optimal salinity regime for *Ruppia tuberosa* in the South Lagoon of the Coorong showing the maximum salinity thresholds for adult plants, seed germination, turion sprouting, turion viability and sexual reproduction.

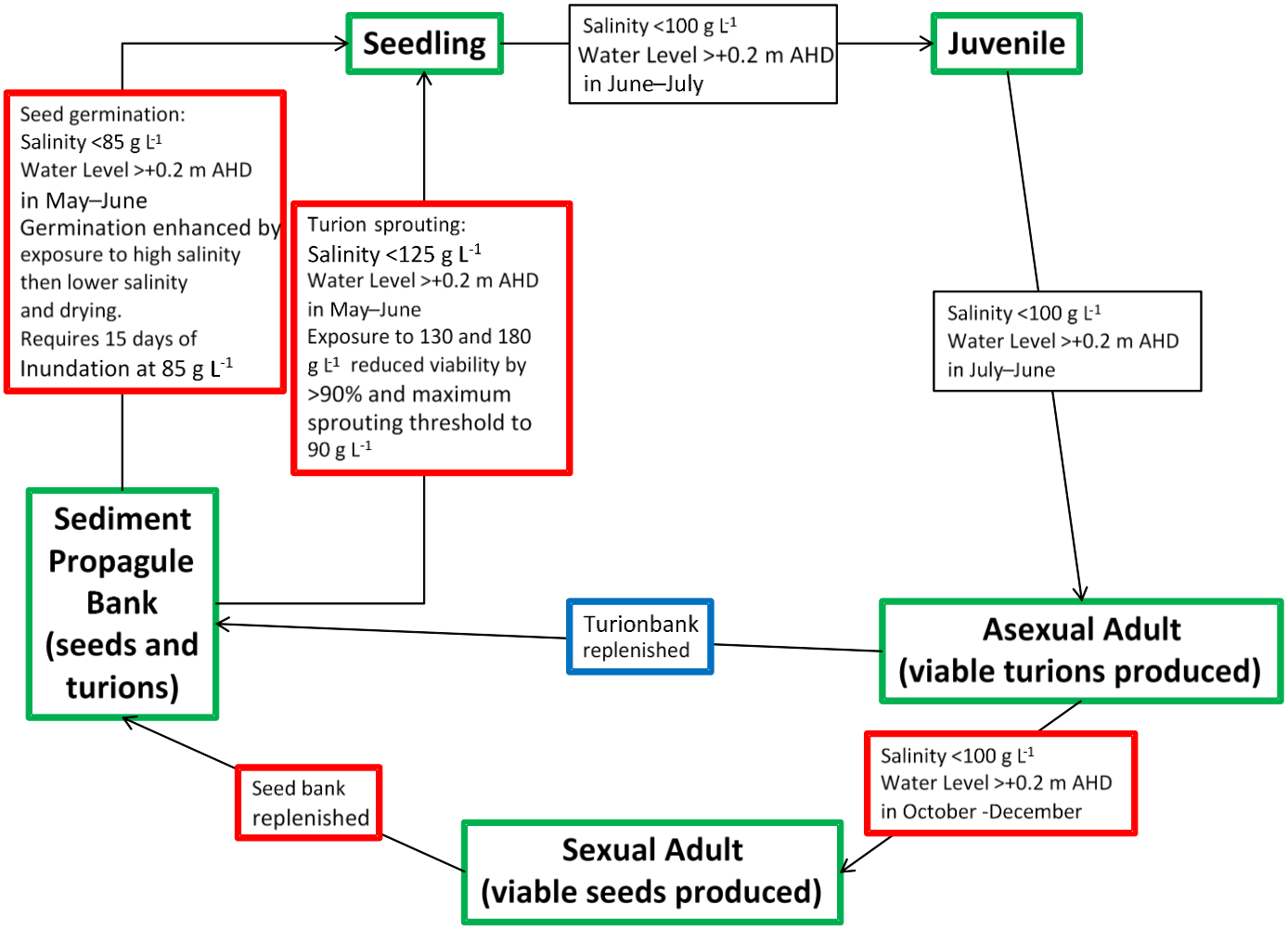


Figure 6. Conceptual model of the life-history of *Ruppia tuberosa* in the South Lagoon of the Coorong. Green boxes represent life-history stages, red boxes potential ecological "bottle necks" and the blue box identifies the need for further study of turion bank replenishment.

Fish habitat

The Coorong supports a diverse range of fish species including freshwater, estuarine, marine and diadromous species. Although having different life-history strategies, many species are strongly associated with estuaries, using them for spawning, nursery and feeding grounds, refuge, or as a migratory pathway (Whitfield 1999). Freshwater inflows and connectivity are crucial for maintaining the habitat, productivity and ecological integrity of estuaries. Over the past years, many studies in the Coorong have identified salinity as the key driver that influences fish assemblage structure (e.g. Noell *et al.* 2009; Zampatti *et al.* 2010) and the extent of estuarine fish habitat in the Coorong (e.g. Geddes 1987; Noell *et al.* 2009; Ye *et al.* 2011b, 2015c). The contraction of effective fish habitat for a range of species due to reduced freshwater inflow and increased salinities was well demonstrated during the millennium drought (2001–2010) (Noell *et al.* 2009; Ferguson *et al.* 2013; Ye *et al.* 2013a). The restoration of fish habitat was further shown in recent high flow years (2010–2013) following the substantial increase in barrage release and a broad salinity reduction throughout the Coorong (Ye *et al.* 2012; Livore *et al.* 2013).

A recent study investigated the tolerance thresholds of the juveniles of key Coorong fish species to hyper-marine salinity and the relationship between thresholds and distribution of species and salinity levels in the field (McNeil *et al.* 2013). These species include the important commercial and recreational fishery species, mulloway (*Argyrosomus japonicus*), yelloweye mullet (*Aldrichetta forsteri*), black bream (*Acanthopagrus butcheri*) and greenback flounder (*Rhombosolea tapirina*), and species with high ecological and conservation values, Tamar goby (*Afurcagobius tamarensis*), congolli (*Pseudaphritis urvillii*), and smallmouthed hardyhead (*Atherinosoma microstoma*).

Gradual acclimation tolerance trials were conducted in aquaria at two different test temperatures; 14 °C (representative of cool 'winter' temperature) and 23 °C (representative of warm 'summer' temperature). The lethal concentrations (i.e. tolerance thresholds) were compared to the distribution of fish across a natural salinity gradient in the field (Noell *et al.* 2009). The results of this study suggest that threshold values, in particular 10% lethal concentration (LC_{10}) and 50% lethal concentration (LC_{50}), can approximate the maximum salinity extent of field

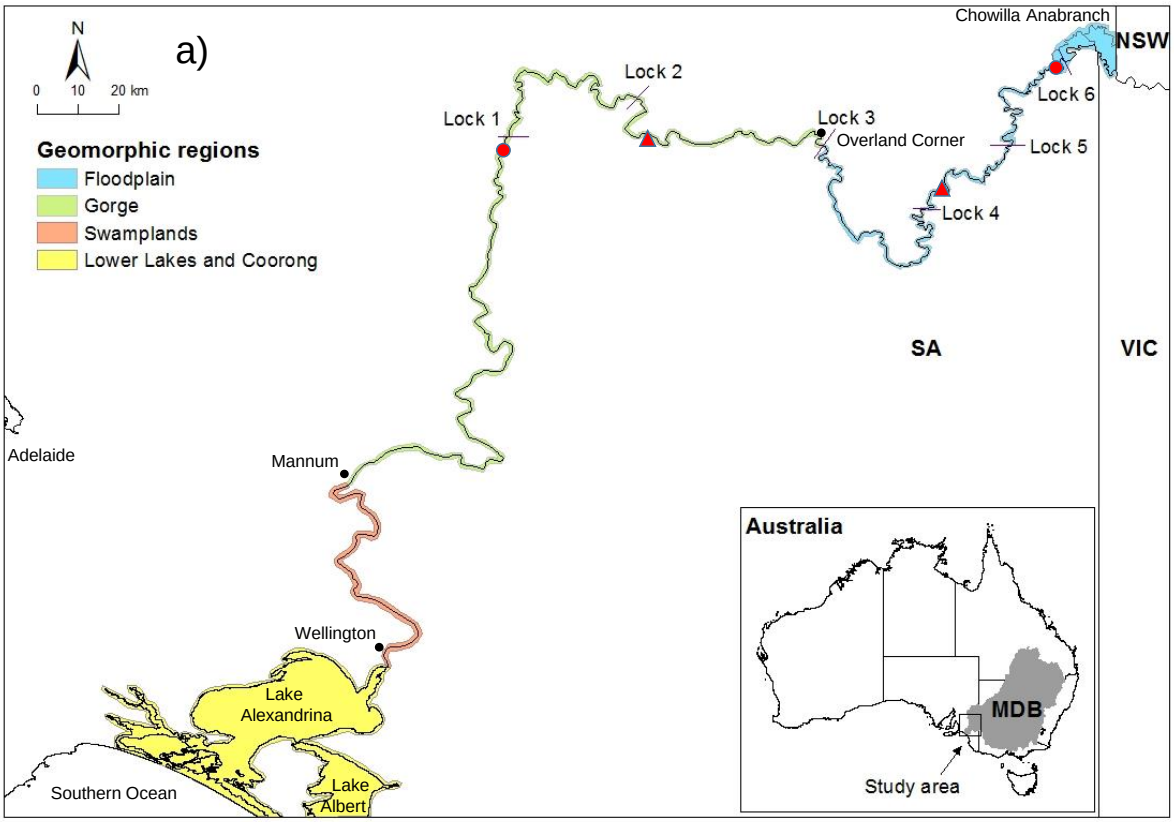
distribution of species with moderate accuracy. As juvenile fish were generally reported to be more sensitive to salinity impacts than adults (Hart *et al*. 1991; Clunie *et al*. 2002), their salinity tolerance thresholds were used in the fish habitat modelling for the Coorong. LC_{10} was adopted as a more conservative threshold than LC_{50} to simulate suitable fish habitat in the field.

3 GENERAL METHODOLOGY

3.1 Study area

This study was conducted at various sites in the main channel and selected wetlands of the Lower Murray River and across the Lower Lakes, Coorong and Murray Mouth region, South Australia (Figure 7). The Lower Murray River (the stretch of the Murray River between the South Australian border and Wellington) covered three distinct geomorphic regions:

- floodplain (South Australian border to Overland Corner);
- gorge (Overland Corner to Mannum);
- swamplands (Mannum to Wellington).



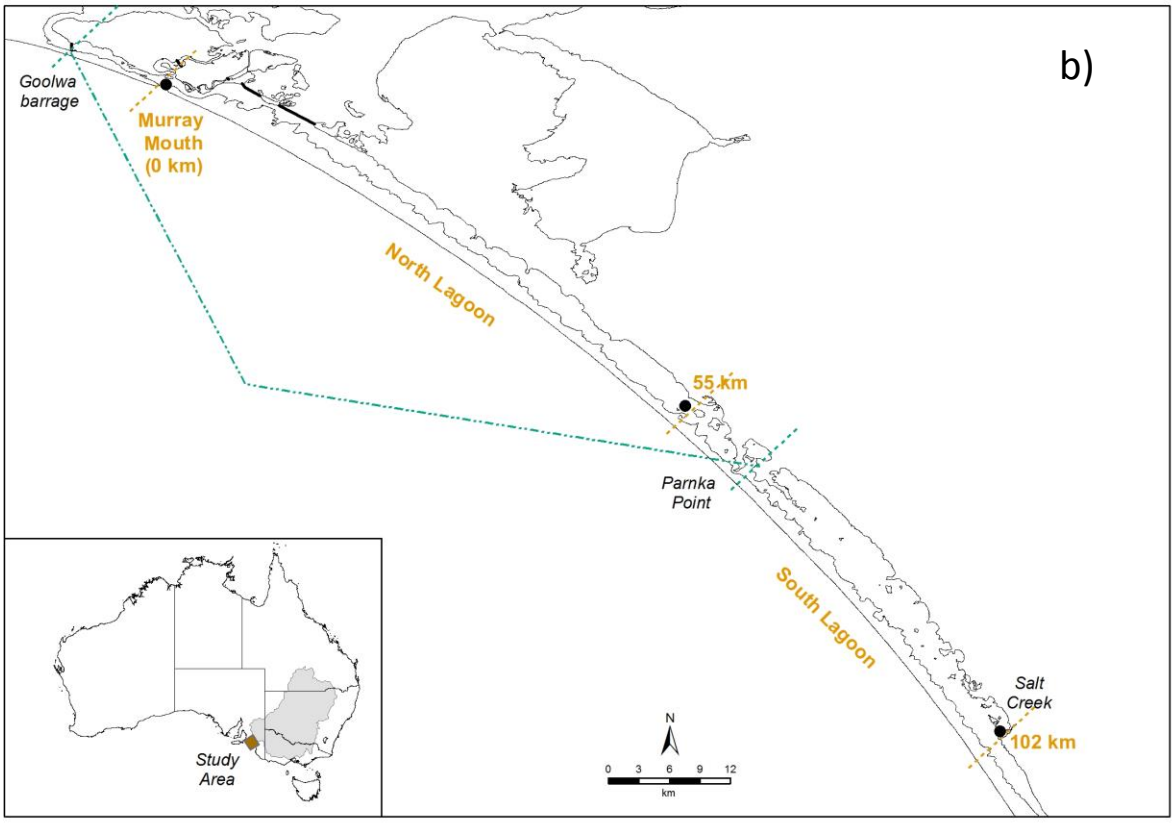


Figure 7. Map of a) the Lower Murray River, Lower Lakes and Murray Mouth showing geomorphic regions; b) the Coorong showing North and South lagoons and 0, 55 and 102 km markers used for Coorong modelling. Red circles represent golden perch larval sampling sites and red triangles represent the wetlands sampled for fish lateral movement.

For the indicator of golden perch spawning and recruitment, the term of 'lower Murray River' was also used, which refers to the stretch of the Murray River downstream of the Darling River junction to Wellington. More detailed information about sampling sites for each monitoring or modelling task is provided below.

Analysis of strontium ratios ($^{87}Sr/^{86}Sr$) in water at sites across the southern MDB

To determine spatio-temporal variation in ratios of strontium ($^{87}Sr/^{86}Sr$) in water over the spring/summer of 2013-14, water samples were collected on a fortnightly–monthly regime from 12 sites across the southern MDB (Table 2; Appendix).

Table 2. Location of water sample collection for $^{87}Sr/^{86}Sr$ analysis.

River	Location	Sampling period	Total number of samples
Murray	Lock 1	16/09/13–17/02/14	12
Murray	Lock 6	17/09/13–04/02/14	11
Murray	Lock 9	01/10/13–21/01/14	9
Murray	Lock 11	23/09/13–03/02/14	10
Murray	Swan Hill	30/09/13–13/02/14	10
Murray	Torrumbarry	29/09/13–30/01/14	5
Murray	Echuca	22/10/13 –16/12/13	5
Murray	Barmah	16/10/13–17/12/13	6
Darling	Weir 32	04/11/13–26/02/14	9
Edward–Wakool	Deniliquin	24/09/13–11/02/14	11
Murrumbidgee	Narrandera	17/09/13–04/02/14	11
Goulburn	Yambuna	07/10/13–17/12/13	6

Sampling golden perch eggs and larvae

Larval fish sampling was conducted at two sites in the Lower Murray River, 5 km downstream of Lock 1 (34°21.138' S, 139°37.061' E) and 5 km downstream of Lock 6 (33°59.725' S, 140°53.152' E) (Figure 7a).

Sampling YOY golden perch and population age-structure

Adult and juvenile golden perch were sampled by boat electrofishing at eight sites in the main channel of the South Australian Murray River and 22 sites in the Chowilla Anabranch system and adjacent Murray River in autumn (March/April) 2014 (Figure 7a; Table 3).

Table 3. Details of boat electrofishing sites in the lower Murray River.

River	Site	Latitude	Longitude
Murray	Murtho Forest	S34.07974	E140.75085
Murray	Plushes Bend	S34.22775	E140.74009
Murray	Rilli Island	S34.39145	E140.59164
Murray	Cobdogla	S34.21724	E140.36522
Murray	Lowbank	S34.16490	E140.03611
Murray	Morgan	S34.05534	E139.68784
Murray	Swan Reach	S34.55317	E139.60809
Murray	Caurnamont	S34.83723	E139.57341

Dissolved and particulate material transport

The study area was in the main channel from Lock 1 (Blanchetown) to the Murray Mouth, incorporating the Lower Murray River, Lower Lakes and Northern Coorong (Figure 7).

Lateral movements of fish

Fish sampling was undertaken at two wetlands in the Lower Murray River (Figure 7a), in the Floodplain region: Martin's Bend (MB) – S 34.176, E 140.371 and Hart Lagoon (HL) – S 34.166, E 139.955.

Ruppia tuberosa and fish habitat

The modelling was conducted for the areas from Murray Mouth, to the North and South lagoons of the Coorong (Figure 7b).

3.2 Methods

Golden perch spawning and recruitment

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

Aliquots (20 ml) 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 studies 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 2M and 7M 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 ≥4,000; no blank corrections were therefore deemed necessary. Strontium isotope analyses were carried out on a "Nu Plasma" multi-collector inductively coupled plasma mass spectrophotometer (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 normalising 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.

Sampling golden perch eggs and larvae

Larval fish sampling was conducted approximately fortnightly between October 2013 and early February 2014. Three day-time and three night-time plankton tows were undertaken on the same day at each site, with both sites sampled within a two-day period. Plankton tows were conducted using a pair of square-framed bongo nets with 500 µm mesh; each net was 0.5 x 0.5 m and 3 m long. Drift nets consisted of 500 µm mesh and were 1.5 m long with a 0.5 m diameter mouth opening. The volume of water (m^3) filtered through each net was determined using a calibrated flow meter (General Oceanics™, model

2030R) placed in the centre of the mouth openings. Fish in all samples were preserved (70-95% ethanol) in the field and returned to the laboratory for processing. Samples were sorted using a dissecting microscope. Larvae and eggs were identified, and where possible, classified as pre-flexion (i.e. early stage larvae with notochord predominately straight) or post-flexion (i.e. the start of upward flexion of the notochord and appearance of fin rays and fin fold) following Serafini and Humphries (2004).

Sampling YOY golden perch and population age-structure

Adult and juvenile golden perch were sampled by boat electrofishing using either a 5 kW or 7.5 kW Smith Root (Model GPP 5 or 7.5) electrofishing unit. Sampling was undertaken in March/April 2014 to maximise the chance of collecting YOY golden perch spawned in the spring–summer 2013-14 spawning season. Electrofishing was conducted during daylight hours and all available littoral habitats were fished. At each site the total time during which electrical current was applied ranged from approximately 1,000 to 1,800 seconds. All fish were measured to the nearest mm (total length, TL) and a subsample, proportionally representing the length-frequency of golden perch collected, were retained for ageing. A subsample of fish (n = 49–67) proportionally representing the length-frequency of golden perch collected from the gorge and floodplain geomorphic regions of the Lower Murray River and the lower Darling River was retained for ageing. Additional juvenile fish were obtained from plankton tows conducted as part of larval fish sampling in December 2013 to February 2014 and from *ad hoc* fyke net sampling in the wetlands connected to the Lower Murray River main channel (floodplain geomorphic region) in December 2013.

Ageing

Larvae and YOY

To estimate the spawn date of larval and YOY golden perch, daily increment counts in otolith microstructure were examined in 38 fish collected from the Lower Murray River. Golden perch larvae/juveniles were measured to the nearest millimetre and sagittal otoliths were removed. Otoliths were mounted

individually in Crystalbond™, proximal surface downwards, and polished down to the primordium using a graded series of wetted lapping films (9, 5, and 3 µm). Sections were then polished using 0.3 µm alumina slurry to a thickness of 50–100 µm.

Sections were examined using a compound microscope (x 600) fitted with a digital camera and Optimas image analysis software (version 6.5, Media Cybernetics, Maryland, USA). 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 two readers for one otolith from each fish. If these differed by more than 10%, or differed by more than 3 days in the case of very young fish (<30 days), 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 (Brown and Wooden 2007) and spawn dates were determined by subtracting the estimated age from the capture date (Zampatti and Leigh 2013b).

Juveniles and adults

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 (n = 176) were euthanized and sagittal otoliths were removed. Whole otoliths were embedded in clear casting resin and a single 400 to 600 µm transverse section was prepared. Sections were examined using a dissecting microscope (x 25) under transmitted light. Estimates of age were determined independently by three readers by counting the number of discernible opaque zones (annuli) from the primordium to the otolith edge. YOY (<1 year old) fish were defined as individuals lacking clearly discernible annuli.

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

Larvae and YOY otolith preparation

Sagittal otoliths of 22 larval and YOY fish collected from the Lower Murray River were dissected and mounted individually in Crystalbond™, proximal surface downwards, on an acid-washed glass slide and polished down to the primordium using a graded series of wetted lapping films (9, 5 and, 3 µm). The slide was then reheated and the polished otolith transferred to a 'master' slide, on which otoliths from all collection sites were combined and arranged randomly to remove any systematic bias during analysis. The samples were rinsed in Milli-Q water (Millipore) and air dried overnight in a class 100 laminar flow cabinet at room temperature.

LA-ICPMS

Laser ablation – inductively coupled plasma mass spectrometry (LA-ICPMS) was used to measure $^{87}Sr/^{86}Sr$ in the otoliths of larval, juvenile and adult fish. 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 a 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 5 or 10 µm sec^{-1} (depending on the size of the otolith) 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 carrier gas (Ar) 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 Rb interferences were made following the procedure 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) which 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.

Lateral movements of fish

Field sampling

Fish assemblages moving through the river-wetland connection passage of two wetlands in the Lower Murray River were sampled to describe the lateral movements of native and exotic fish before, during and after the delivery of the environmental water in 2013-14. Sampling was undertaken during four sampling events (SE) which corresponded to four phases of flow delivery (Figure 8) in order to examine the influence of environmental water delivery and associated hydrological cues on fish movement to and from wetlands. The four flow phases were: (SE1) Stable – unchanging flows; (SE2) Receding – regulated recession of flow; (SE3) Rising EW – increasing flow associated with the delivery of the environmental water; and (SE4) Receding EW – recession of flow associated with environmental water delivery.

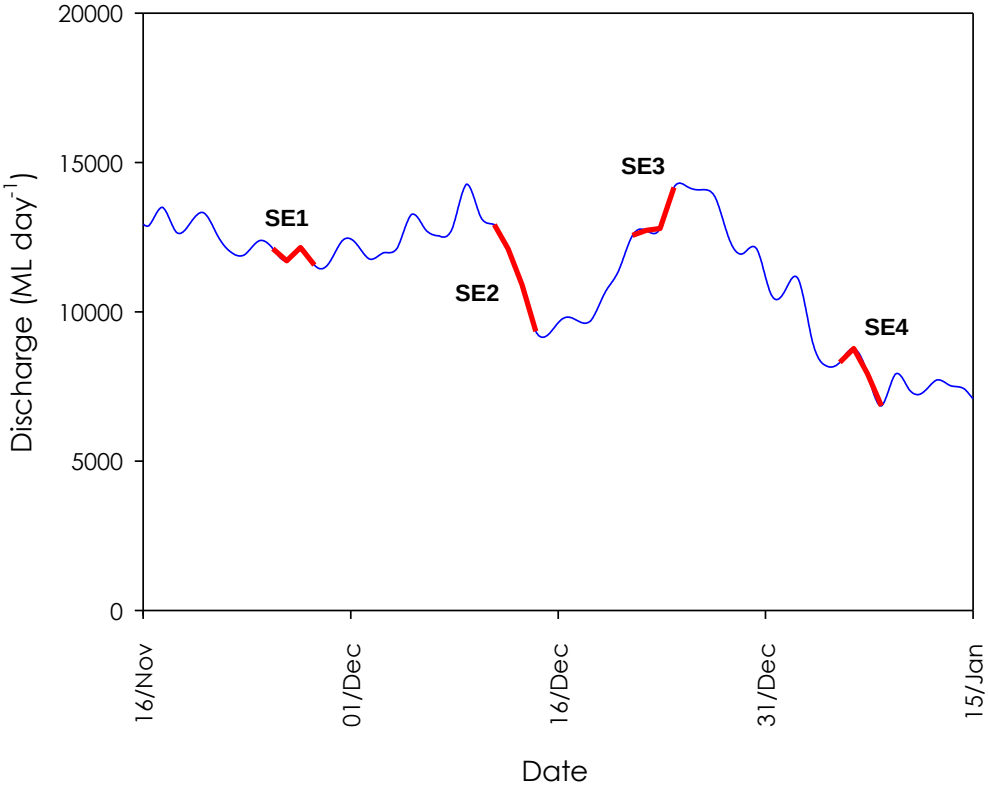


Figure 8. Hydrograph showing the Murray River flow (ML day^{-1}) across the border to South Australia for the period November 2013 to January 2014. Red, numbered (SE1–4) lines indicate the four phases of flow during which sampling were undertaken.

On each sampling occasion, a network of fyke nets and drum nets were set in the river-wetland connection passage of each wetland, to sample fish moving bi-directionally between the main river channel and the wetland. All nets were set for 24 hours on three consecutive days (replicates). All captured fish were identified to species level, and counted. Up to 30 individuals of each species were randomly sub-sampled and measured for total length (TL mm) and released away (approximately 50 m) from the net fleet in the direction they were originally moving. Native fish species were assigned to one of five functional groups based on spawning strategy, flow preferences and life-cycle characteristics. Each species was categorised as: long-lived apex predator, flow-dependent specialist, foraging generalist, floodplain specialist, or diadramous (Baumgartner *et al.* 2013). All exotic species were categorised into a single group.

On each sampling occasion, measurements of pH, dissolved oxygen, temperature and conductivity were recorded. Data for flow discharge, relative

water level (m AHD) were obtained for both wetlands from the DEWNR Surface Water Archive (www.waterconnect.sa.gov.au; accessed on 16/05/2014).

Data analysis

For data analyses, fish abundance data were standardised for catch-per-unit-effort (CPUE) to provide a relative abundance (no. of fish 24 hours^{-1}). Fish assemblage structure was defined by the species composition and the relative abundance of each species within each replicate (day) over each flow phase for each wetland. Spearman rank correlations were used to detect correlations between environmental variables and where such correlations were detected, one variable was removed from subsequent analyses. Multivariate statistical analyses were undertaken using the PRIMER v6 package (Clarke and Warwick 2001) to investigate: (1) differences in assemblage structure moving 'In' and 'Out' of each wetland between and across flow phases; (2) what are the key species that drive the differences between assemblages; and (3) whether variation in fish assemblage structure was correlated to changes in hydrology. The changes in individual species abundances throughout the flow phases in a given wetland were assessed using one-way and two-way ANOVAs for assemblages within and moving in/out of wetlands, respectively. Assumptions of normality and homogeneity of variance were tested using the square-root transformed data and species that did not meet these assumptions after were not included in subsequent analyses.

Dissolved and particulate material transport

Water quality was monitored between July 2013 and June 2014 (Appendix 1). At each sampling site, measurements of water temperature, electrical conductivity, dissolved oxygen, pH and turbidity were taken at 0.5 m intervals through the water column. In addition, integrated-depth water samples were collected and sent to the Australian Water Quality Centre, an accredited laboratory of the National Association of Testing Authorities. Samples were analysed for filterable reactive phosphorus (herein called phosphate), total phosphorus, nitrate, ammonium, total Kjeldahl nitrogen (the sum of organic nitrogen and ammonium), dissolved silica, total suspended solids, suspended

organic matter and chlorophyll *a* using standard techniques. Organic nitrogen was calculated as the difference between total Kjeldahl nitrogen and nitrate.

The physico-chemical information was used to validate a three-dimensional hydrodynamic-biogeochemical model, TUFLOW-FV-AED (see Appendix 1). Unlike, the model used for previous evaluations (Aldridge *et al*. 2013; Ye *et al*. 2014) this model has a single modelling domain and TUFLOW-FV is now used extensively in the region for hydrological purposes. The model was initialised with data from several data sources, including Lock 1 inflows with the different combinations of environmental water use (provided by the MDBA). The flow data are treated as indicative only due to complexities around differences in interstate water accounting. Assumptions such as these, result in uncertainty in the model outputs and so outputs are not be treated as absolute values (refer to Aldridge *et al.* 2013 for more detail). When assessing the relative differences between scenarios, the uncertainties are considered to influence the accuracy of each scenario equally and so the model outputs are used to assess the general response to environmental watering.

For this study, four simulations were run and compared for 1 July 2013 to 30 June 2014:

- with both Commonwealth environmental water and TLM water;
- with Commonwealth environmental water, but without TLM water;
- without Commonwealth environmental water, but with TLM water;
- without both Commonwealth environmental water and TLM water.

The influence of environmental watering on the concentrations of water-borne matter was assessed through a comparison of modelled concentrations for the Lower Murray River (Wellington), Lower Lakes (Lake Alexandrina Middle) and Murray Mouth. The transport of matter was assessed through modelled exports from the Lower Murray River (Wellington), Lower Lakes (Barrages) and Murray Mouth. Findings are presented for salinity, ammonium, phosphate, dissolved silica, organic nitrogen, organic phosphorus, chlorophyll *a* and total suspended solids. Salinity is presented as practical salinity units (PSU), a measurement of the measured conductivity to standard KCl conductivity. PSU was used for

validating model outputs as it overcomes observed differences in electrical conductivity caused by changes in water temperature. One PSU is approximately equal to part per thousand.

Coorong modelling

Hydrodynamic model

The hydrodynamic model, *Coorong Hydrodynamic Model* (CHM) v2.1, developed by CSIRO (Webster 2007; Webster 2013; Joehnk and Webster 2014) was used to simulate water level and salinity along the Coorong for 102 km from the Murray Mouth to the southern end of the South Lagoon (e.g. Joehnk and Webster 2014). The hydrodynamic model is based on daily barrage flow values from 1963 and updated regularly, currently until 30 June 2014 (data provided by MDBA). Salinities and water level along the North and South Lagoon were calculated with a daily time step with 1 km resolution driven with hourly and daily datasets describing tidal forcing, wind velocity, evaporation and precipitation. Time series for these fields were updated until June 2014 with tidal data provided by the National Tidal Unit (NTU BoM), evaporation and precipitation downloaded from SILO database, Upper South East Drainage (USED) flows and conductivities and wind data provided by DEWNR (SA). Barrage flows were provided by MDBA including estimates of the percentage of environmental water included in the barrage flow on a monthly basis. Flow scenarios were constructed to account for the possible absence of such water. Data available for this split is for July 2011 to June 2014. All new data were error checked and gaps were closed where necessary before adding them to the CHM database. The model was run to simulate the whole period from 1963 to 2014.

Additionally, to evaluate the effect of environmental watering on ecosystems, three scenarios were calculated for the 2013-14 watering year:

- reference, including all water sources;
- with Commonwealth environmental water, without TLM water;
- without both Commonwealth environmental water and TLM water.

The results of salinities and water levels from scenario runs were subsequently used to simulate habitat characteristics for *Ruppia tuberosa* as well as fish species.

Ruppia tuberosa

Based on the effect of salinity and water level on the life-history of *R. tuberosa*, an ecological response model that calculates the probability of replenishing the sediment propagule bank based on modelled hydrological conditions (i.e. output of the CHM) was developed. For the purpose of modelling, each life-history stage was treated as occurring over a discrete time period (in nature there would be considerable overlap of life-history stages) and a survival probability calculated for each time block that was in turn used to calculate the probability of replenishing the sediment propagule bank. As this model is based on thresholds, simulation results can include jumps from one state to the next one. This need to be interpreted not as a jump *per se*, but as having surpassed a threshold.

Model processes include seed germination, turion sprouting, seedling development to juvenile plants, juvenile development to asexual adult plants, and asexual adult development to sexual adult plants. Detailed modelling descriptions are provided in Joehnk *et al.* (2014). Each process is assigned a survival probability, which in the end results in a combined probability of sediment propagule bank replenishment.

Fish habitat

The basic fish model analysed salinity thresholds for the juveniles of key fish species on an annual ("habitat suitability") and daily basis. The annual analysis provided probabilities of habitat suitability by calculating the annual exceedance probability for a certain salinity threshold. A simpler approach only detected the extent of a certain salinity threshold on a daily basis ("threshold analysis") along the Coorong starting at the Murray Mouth. The latter basically gives a simplified picture of the salinity contours along the Coorong over time.

Annual habitat suitability

The aim of the fish model was to identify suitable habitats for juveniles of different species for the region from the Murray Mouth to the end of the South Lagoon of the Coorong, depending on ambient salinity levels for a period of one watering year. Each fish species was characterised by a certain salinity tolerance threshold, which varied between cool and warm seasons (Table 4). In our simplified expert model type system, the cool months included the period from mid April to mid October and warm months included the period from mid October to mid April. The model effectively calculated an annual exceedance probability for salinity. It assessed the daily salinity output of the hydrodynamic model for each 1 km grid along the Coorong, and counted the number of days where salinity was below the threshold of a particular fish species during the cool or warm period. The annual probability that a 1 km grid was a suitable habitat was then calculated. Details of parameter definition and model programming are presented in Joehnk *et al.* (2014).

Table 4. Period specific salinity thresholds for suitable habitats for seven fish species (adapted from McNeil *et al.* 2013).

Species name	Threshold cool months [g L^{-1}]	Threshold warm months [g L^{-1}]
Mulloway	60.3	51.1
Tamar goby	67.7	66.3
Black bream	78.6	81.8
Greenback flounder	81.1	72.9
Yelloweye mullet	83.8	68.3
Congolli	89.5	86.9
Smallmouthed hardyhead	99.5	97.1

Daily fish salinity threshold analysis

To create an overview of suitable habitats on a daily basis, the distance from the mouth (or the number of 1 km wide cells in the underlying hydrodynamic model) to which a certain threshold of salinity was not surpassed was also calculated. This was achieved via a simplified contour plot depicting regions below and above a certain salinity threshold (see e.g. Figure 37).

4 FINDINGS

4.1 Golden perch spawning and recruitment

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

Collection of water samples commenced in late September/early October 2013 and extended through February 2014 at most sites. Overall, $^{87}Sr/^{86}Sr$ at most locations remained reasonably stable throughout the period of collection, with the highest ratios (>0.7190) measured in the Murray River at Barmah and the Edward River, and the lowest (<0.7080) in the Darling River (Figure **9**). Water $^{87}Sr/^{86}Sr$ generally decreased longitudinally along the Murray River where tributaries with distinct and temporally stable $^{87}Sr/^{86}Sr$ (e.g. Goulburn and Murrumbidgee rivers) contribute to discharge. At sites in the lower Murray River (downstream of the Darling River junction), $^{87}Sr/^{86}Sr$ was highly variable due to substantial temporal variation in water source (i.e. from the mid Murray, Darling River and Lake Victoria), although $^{87}Sr/^{86}Sr$ at Lock 1 remained reasonably stable (Figure **9**).

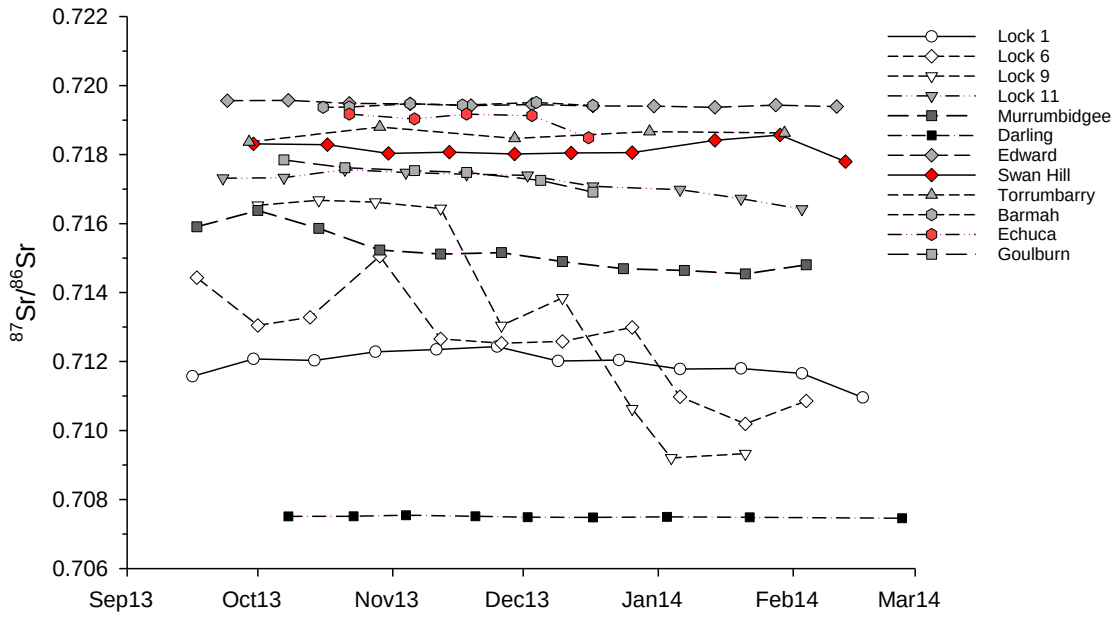


Figure 9. $^{87}Sr/^{86}Sr$ ratios in water samples collected from mid-September/early October 2013 to February 2014 in the Murray River (Lock 1, 6, 9, 11, Swan Hill, Torrumbarry, Echuca and Barmah), and the Darling, Goulburn, Edward and Murrumbidgee rivers.

Lower Murray River and Darling River

From August 2013 to early January 2014, flow in the lower Murray River (discharge at the South Australian border, QSA) generally reflected flow in the mid-reaches of the Murray at Euston. In both regions, flows increased from approximately 8,000 ML day^{-1} in mid-August 2013 to a peak of approximately 25,000–27,000 ML day^{-1} in late September/mid-October 2013. Flows then gradually decreased to approximately 8,000 ML day^{-1} in early January 2014, with some variation in the descending limb of the hydrograph in December 2013 (Figure 10). In early January, flow in the Murray River at Euston decreased to approximately 5,000 ML day^{-1} while flow in the Lower Murray ranged 7,000–10,000 ML day^{-1} as a result of flow from the Darling and water released from Lake Victoria (Figure 10). Flow in the Darling River at Burtundy was <1,000 ML day^{-1} until early December 2013 when flow increased to a maximum of approximately 3,000 ML day^{-1} through late December and early January before gradually decreasing to <1,000 ML day^{-1} by late March 2014 (Figure 10).

From early November 2013 to early January 2014, the daily contribution of Commonwealth environmental water to flow at the South Australian border ranged ~2,300–7,300 ML, with Commonwealth environmental water peaking at ~7,100 ML day^{-1} on 9 December 2013 and at ~7,000–7,300 ML day^{-1} through 24–27 December 2013 (Figure 2). TLM environmental water was delivered from 25 October to 28 December 2013, peaking at ~4,000–4,400 ML day^{-1} through 8–10 November 2013 (Figure 2). Details on the physical sources of environmental water (e.g. Lake Victoria, mid-Murray River or Darling River) remain unclear.

$^{87}Sr/^{86}Sr$ in water samples collected from Locks 9 and 6 in the lower Murray River was highly variable, exhibiting a trend from higher ratios (0.7140–0.7170), representative of the mid-Murray, from October to mid-November 2013, and decreasing ratios after December 2013 resulting from increasing flows from the Darling River and declining flow from the mid Murray (Figure 10). From September 2013 to February 2014, $^{87}Sr/^{86}Sr$ in water samples collected from Lock 1 remained relatively stable ranging 0.7110–0.7125 and from October 2013 to February 2014, $^{87}Sr/^{86}Sr$ in the Darling River at Menindee (Weir 32) remained constant at approximately 0.7075 (Figure 10).

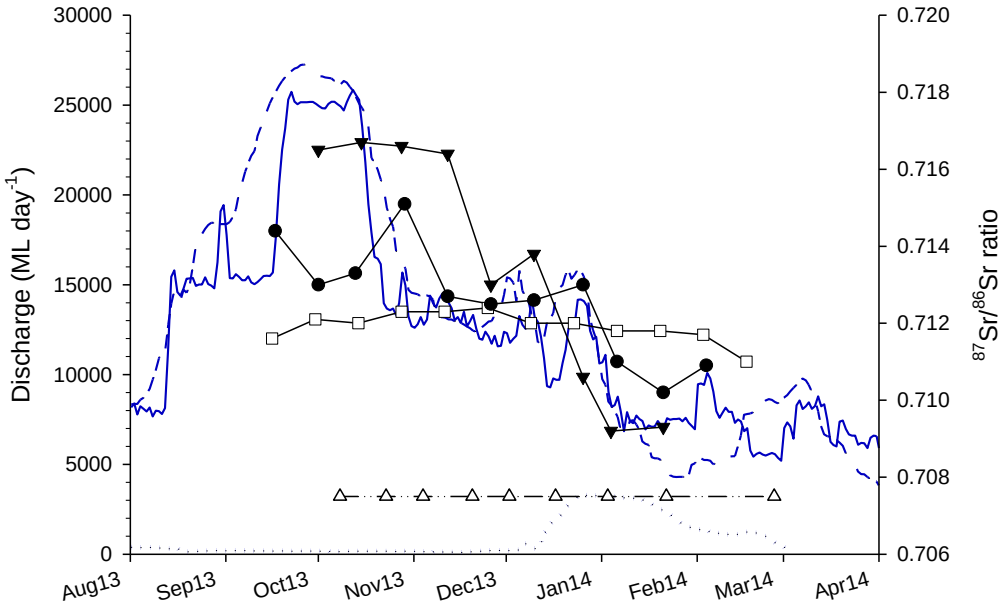


Figure 10. Mean daily discharge (ML day^{-1}) in the Murray River at the South Australian border (solid blue line) and Euston (dashed blue line), and Darling River at Burtundy (dotted blue line). $^{87}Sr/^{86}Sr$ in water samples collected from mid-September 2013 to late-February 2014 in the Murray River at Lock 9 (solid triangles), Lock 6 (solid circles) and Lock 1 (open squares), and the Darling River at Menindee (Weir 32) (open triangles).

Eggs and larvae of golden perch

A total of 123 and 31 golden perch larvae were collected at Lock 1 and Lock 6, respectively, in 2013-14. At Lock 1, larvae were first collected in early October when the water temperature reached ~19 °C. Larval abundance peaked in mid-December 2013 and larvae continued to be collected until early February 2014 (Figure 11) with all larvae from Lock 1 at the pre-flexion stage. At Lock 6, golden perch larvae were collected from early October 2013 to mid-January 2014, with the majority being post-flexion. Larval abundance peaked in mid-October and again in late December 2013 with larvae present until mid-January (Figure 11).

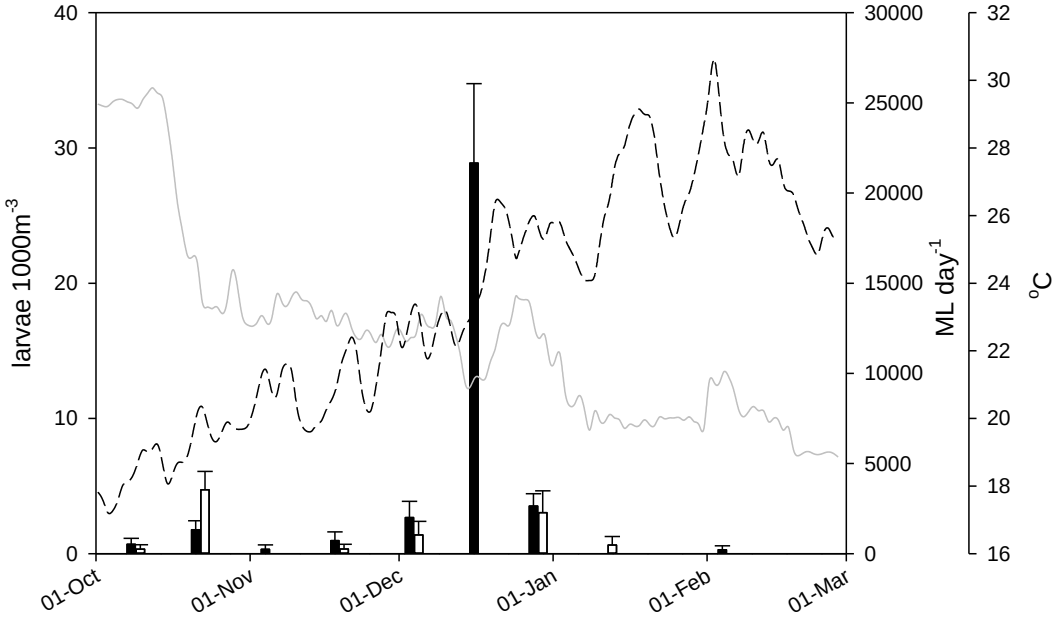


Figure 11. Mean (±S.E.) standardised abundance of golden perch larvae collected in the Murray River at Lock 1 (solid black bars) and Lock 6 (open bars) in 2013-14, plotted against discharge (ML day^{-1}) in the Murray River at the South Australian border (solid grey line) and water temperature (°C) (dashed black line). Sampling was undertaken fortnightly from 9 October 2013 to 5 February 2014.

Spawn dates and otolith $^{87}Sr/^{86}Sr$ of larval and YOY golden perch

In 2013-14, daily ages and spawn dates were estimated for 38 larval and YOY golden perch collected from the Lower Murray River. Ages ranged from 4–115 days for fish collected from 4 December 2013 to 4 March 2014 indicating a spawning period from 14 October 2013 to 24 December 2013 (Figure 11; Figure 12).

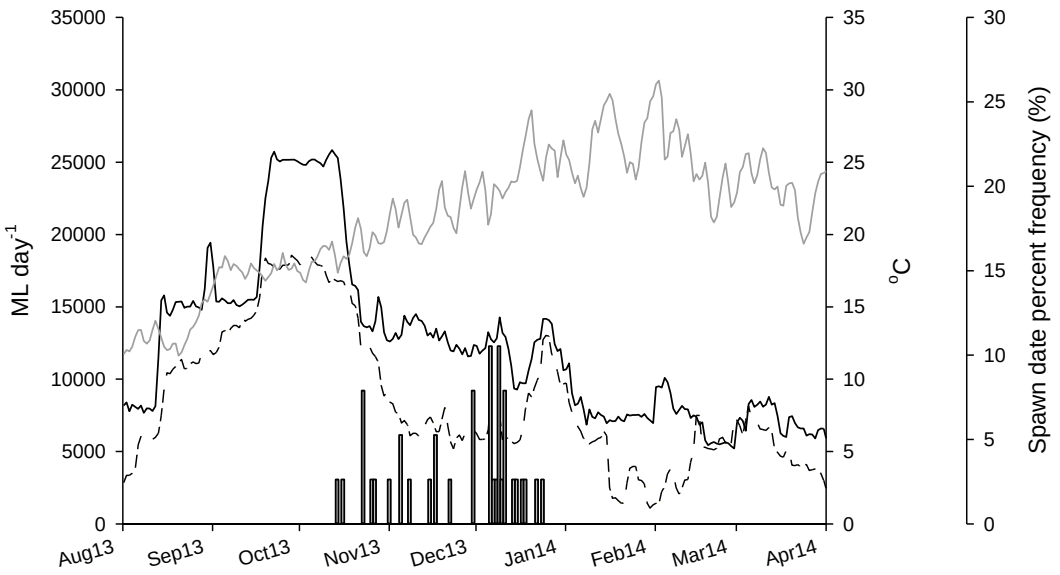


Figure 12. Back-calculated spawn dates for larval and young-of-year golden perch (grey bars; n = 38) captured from the Lower Murray River during 2013-14, plotted against discharge (ML day^{-1}) in the Murray River at the South Australian border (solid black line) and Euston (dashed black line) and water temperature (°C) (grey line).

Pre-flexion larvae collected in larval tows at Lock 1 in December 2013 ranged from 4–7 mm and 4–12 days old (Table 5). For larval tows at Lock 6, larvae and early juveniles from October 2013 to January 2014 ranged from 9–27 mm and from 5–43 days old (Table 5). Juvenile golden perch collected by electrofishing and fyke netting in December 2013 and March 2014, at locations throughout the floodplain geomorphic region of the lower Murray, ranged from 21–53 mm and 37–115 days old (Table 5).

Of the 38 larvae/YOY for which we could determine daily age, otoliths from 18 individuals were analysed for $^{87}Sr/^{86}Sr$ (Table 5). Seventeen age 0+ fish, spawned between 23 October and 24 December 2013, had otolith core $^{87}Sr/^{86}Sr$ indicative of the lower Murray River, downstream of Lock 9 (i.e. >0.7090 and <0.7170) (Table 5; Figure 13). One fish, collected in Salt Creek at Chowilla on 20 March 2014 was spawned on the 14 December 2013 and exhibited otolith core $^{87}Sr/^{86}Sr$ indicative of the Darling River (~0.7075) (Figure 13).

Table 5. Capture location and date, length (mm), age (days) and spawn date of 38 larval/young-of-year golden perch collected from the floodplain and gorge geomorphic regions of the Lower Murray River, including otolith core $^{87}Sr/^{86}Sr$ values measured from 18 larval/young-of-year fish.

Region	Capture location	Capture date	Length (mm)	Age (days)	Spawn date	$^{87}Sr/^{86}Sr$
Floodplain	Lock 6	13/01/2014	27	43	30/11/2013	0.7136725
Floodplain	Lock 6	21/10/2013	11	7	14/10/2013	-
Floodplain	Lock 6	21/10/2013	9	5	16/10/2013	-
Floodplain	Lock 6	30/12/2013	18	23	6/12/2013	0.7138749
Floodplain	Lock 6	30/12/2013	19	24	6/12/2013	0.7134529
Floodplain	Lock 6	12/01/2014	12	20	22/12/2013	-
Floodplain	Lock 6	30/12/2013	20	23	7/12/2013	-
Floodplain	Lock 6	3/12/2013	14	16	17/11/2013	0.7163424
Floodplain	Lock 6	3/12/2013	11	15	17/11/2013	0.7135185
Floodplain	Lock 6	30/12/213	18	24	6/12/2013	0.7138883
Floodplain	Lock 6	12/01/2014	13	19	24/12/2013	0.7109106
Floodplain	Lock 6	30/12/2013	16	21	9/12/2013	0.7134128
Floodplain	Lock 6	30/12/2013	20	24	6/12/2013	0.7138498
Floodplain	Lock 6	30/12/2013	13	20	10/12/2013	-
Floodplain	Lock 6	30/12/2013	15	21	9/12/2013	-
Floodplain	Lock 6	30/12/2013	9	12	18/12/2013	-
Floodplain	Martin's Bend	12/12/2013	22	37	5/11/2013	0.7143233
Floodplain	Martin's Bend	13/12/2013	23	51	23/10/2013	0.7127532
Floodplain	Martin's Bend	13/12/2013	22	38	5/11/2013	0.7162591
Floodplain	Martin's Bend	13/12/2013	23	42	1/11/2013	0.7127598
Floodplain	Martin's Bend	12/12/2013	30	49	23/10/2013	0.7130439
Floodplain	Martin's Bend	12/12/2013	25	47	26/10/2013	-
Floodplain	Martin's Bend	12/12/2013	24	50	23/10/2013	-
Floodplain	Martin's Bend	13/12/2013	21	47	27/10/2013	-
Floodplain	Martin's Bend	22/12/2013	22	43	8/11/2013	-
Floodplain	Monoman Creek	11/03/2014	53	115	15/11/2013	0.7131053
Floodplain	Salt Creek	20/03/2014	43	96	14/12/2013	0.7076807
Floodplain	Border Cliffs	4/03/2014	41	101	22/11/2013	0.7125680
Floodplain	Chowilla Creek	15/03/2014	52	90	15/12/2013	0.7111782

Region	Capture location	Capture date	Length (mm)	Age (days)	Spawn date	$^{87}Sr/^{86}Sr$
Gorge	Lock 1	4/12/2013	5	4	30/11/2013	-
Gorge	Lock 1	4/12/2013	5	4	30/11/2013	-
Gorge	Lock 1	17/12/2013	6	8	8/12/2013	-
Gorge	Lock 1	17/12/2013	6	7	9/12/2013	-
Gorge	Lock 1	17/12/2013	4	6	11/12/2013	-
Gorge	Lock 1	29/12/2013	7	12	17/12/2013	-
Gorge	Lock 1	17/12/2013	6	7	9/12/2013	-
Gorge	Lock 1	17/12/2013	5	6	11/12/2013	-
Gorge	Lock 1	17/12/2013	5	6	11/12/2013	-

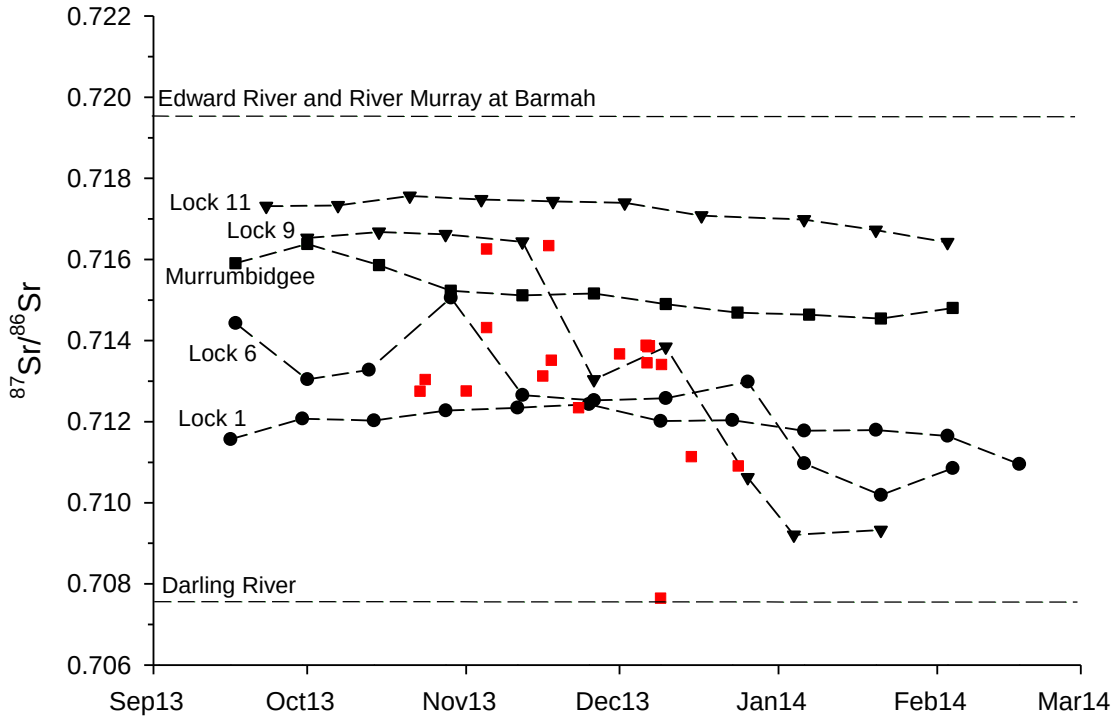


Figure 13.$^{87}Sr/^{86}Sr$ in water samples collected from late September/early October 2013 to February 2014 at sites in the southern MDB. $^{87}Sr/^{86}Sr$ in the Darling River and Edward River/Murray River at Barmah are presented as dashed straight lines as these were temporally stable and represent the maximum and minimum $^{87}Sr/^{86}Sr$ measured in water samples in the southern MDB in 2013-14. Closed red squares represent spawn date and otolith core $^{87}Sr/^{86}Sr$ of larval/YOY golden perch (n = 18) collected in the lower Murray River from October 2013 to April 2014.

Transects of $^{87}Sr/^{86}Sr$ from the otolith core to edge can provide information on the movement history of early life stage golden perch but may also reflect temporal variability in ambient $^{87}Sr/^{86}Sr$ in water. Here we provided preliminary

examples of life-history profiles for two YOY golden perch, based on transects of $^{87}Sr/^{86}Sr$ from the otolith core to the edge. Transects of $^{87}Sr/^{86}Sr$ show that two fish, captured at Chowilla (Lock 6) and Martin's Bend (upstream of Lock 4) in the Lower Murray River, were spawned in the Darling River and Murray River, respectively. Both fish exhibit modulation of $^{87}Sr/^{86}Sr$ as they move downstream (passively and/or actively) (Figure 14). An individual fish spawned in the Darling River in December 2013, and captured upstream of Lock 6 in early March 2014, showed an increase in $^{87}Sr/^{86}Sr$ as it transitioned into the lower Murray River (Figure 14). In contrast, an individual spawned in the lower Murray River in early November 2013, and captured upstream of Lock 4 in mid-December 2013, showed decreasing $^{87}Sr/^{86}Sr$ across the otolith transect (Figure 14). Whilst this fish may have been moving downstream in the lower Murray, the change in otolith$^{87}Sr/^{86}Sr$ was most likely due to dissolved $^{87}Sr/^{86}Sr$ in the lower Murray River decreasing substantially over this period due to increasing Darling River flow (Figure 10).

Otolith core $^{87}Sr/^{86}Sr$ was also determined for four age 0+ golden perch that were collected from the floodplain geomorphic region of the lower Murray River in April 2014 but for which spawn dates were unable to be accurately determined. The fish collected in the lower Murray River exhibited core $^{87}Sr/^{86}Sr$ values consistent with a lower Murray River origin (i.e. >0.7080 and <0.7170) (Table 6).

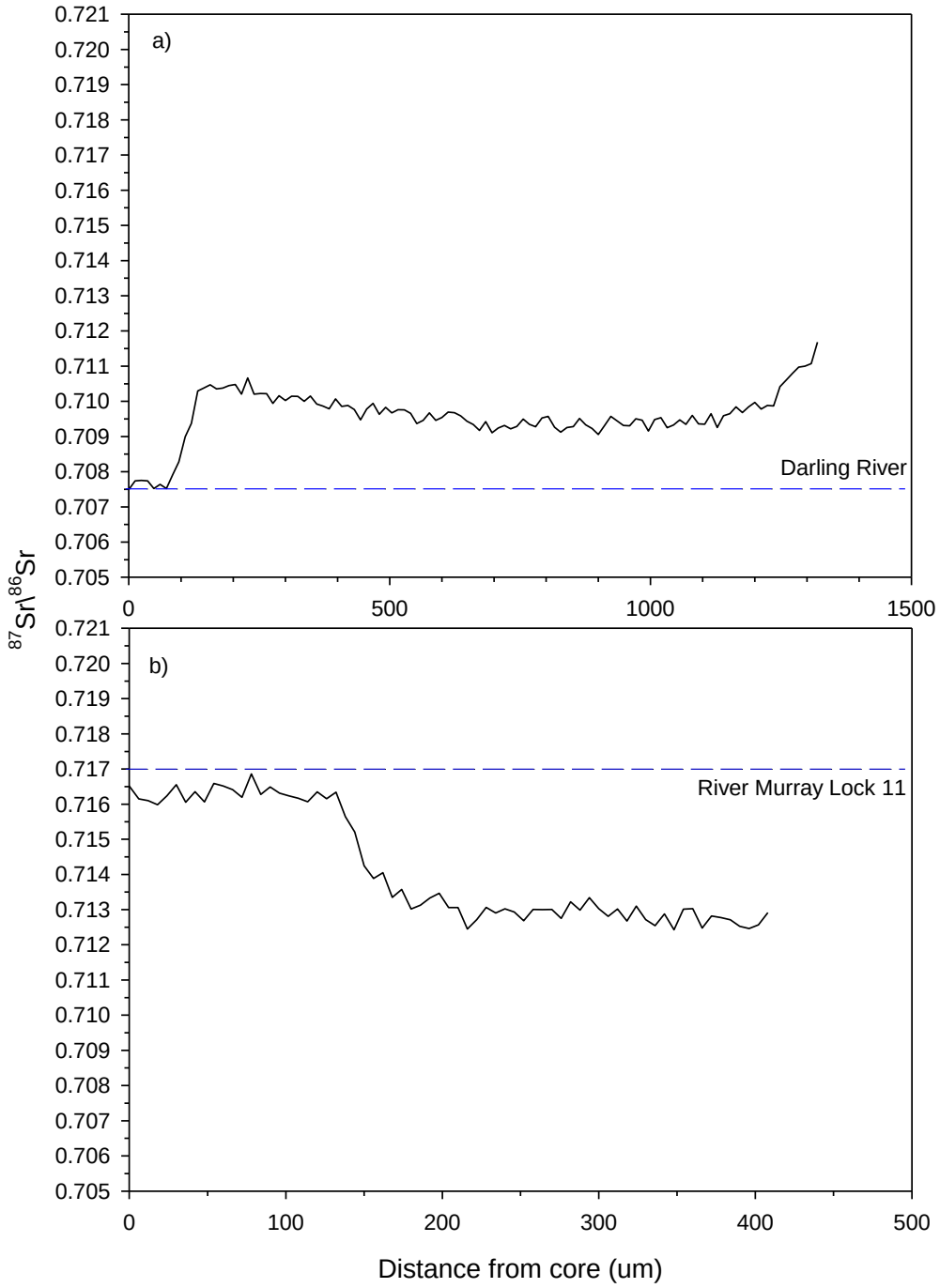


Figure 14. Individual life history profiles based on otolith Sr isotope transects (core to edge) for two juvenile golden perch aged (a) 101 and (b) 38 days collected at Chowilla and Martin's Bend (upstream of Lock 4) respectively in the floodplain region of the lower Murray River. Dashed blue line denotes $^{87}Sr/^{86}Sr$ ratio in (a) the Darling River and (b) Murray River at Lock 11.

Table 6. Otolith core $^{87}Sr/^{86}Sr$ measured in 4 young-of-year golden perch collected from the floodplain geomorphic region of the lower Murray River in April 2014.

Region	Capture location	Capture date	Length (mm)	Age (yrs)	Spawned	$^{87}Sr/^{86}Sr$
Floodplain	Chowilla Ck	6/03/14	42	0+	2013-14	0.7104077
Floodplain	Salt Ck (Chowilla)	20/03/14	63	0+	2013-14	0.7150462
Floodplain	Downstream Lock 4	30/04/14	63	0+	2013-14	0.7118915
Floodplain	Downstream Lock 4	30/04/14	80	0+	2013-14	0.7096538

Length and age structure of golden perch

In 2014, golden perch sampled in the gorge and floodplain geomorphic regions of the lower Murray River (including the Chowilla anabranch system) ranged in age from 0+ to 18+ years, with a dominant cohort of age 4+ fish, spawned in 2009-10, comprising 52%, 64% and 45% of the sampled population in the Chowilla, floodplain and gorge geomorphic regions, respectively (Figure 15). The second most abundant cohort in Chowilla, and the broader floodplain geomorphic region of the lower Murray River, was age 3+ fish spawned in 2010-11 (Figure 15). In the gorge geomorphic region, the second most abundant cohorts were age 3+ and 13+ fish spawned in 2010-11 and 2001-02 and comprising 18% and 19%, respectively, of the sampled population (Figure 15). Age 0+ fish, spawned in 2013-14, comprised ≤5% of the sampled populations in the floodplain (including Chowilla) and gorge geomorphic regions of the lower Murray (Figure 15).

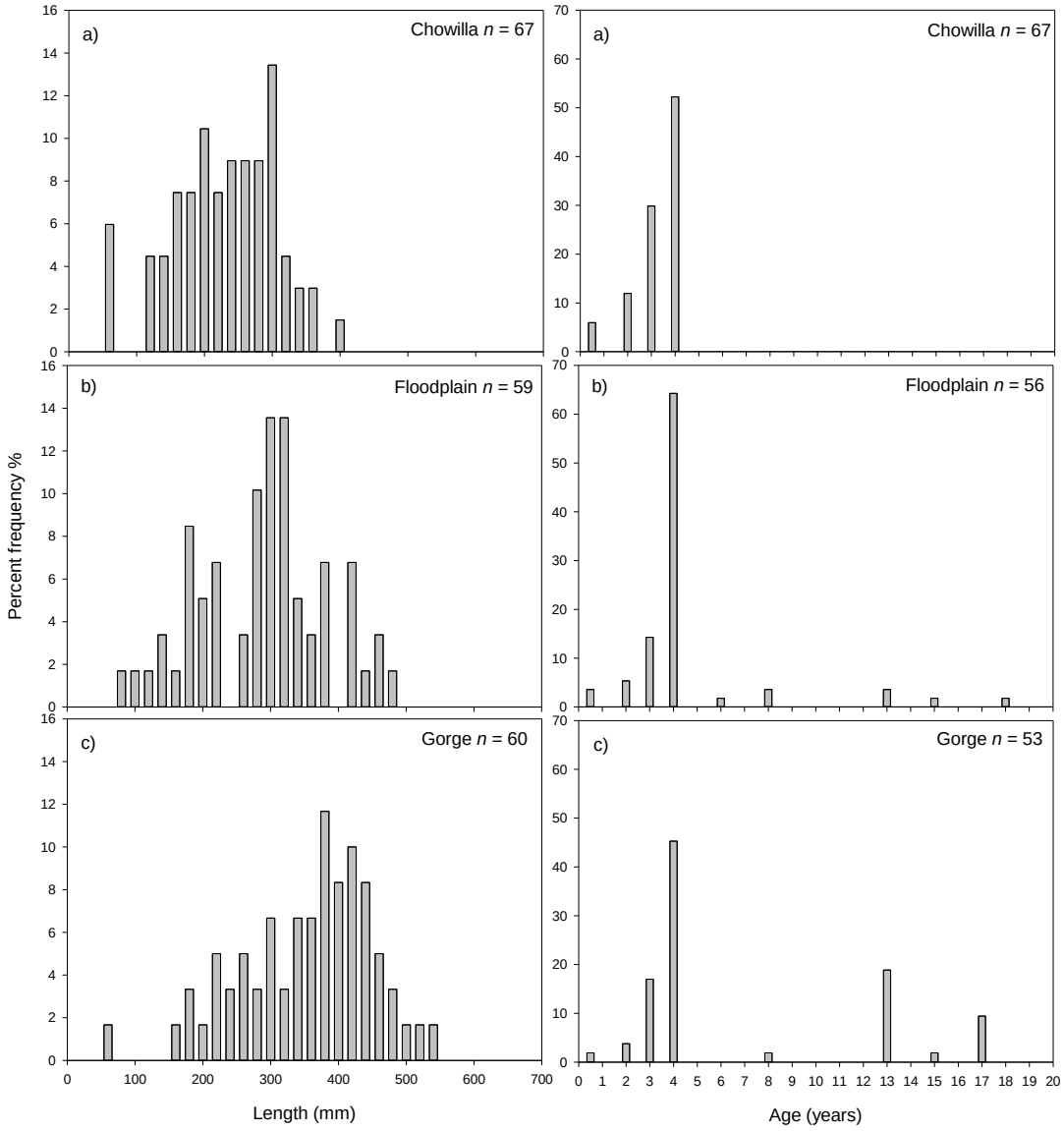


Figure 15. Length (left column) and age (right column) frequency distribution of golden perch collected by boat electrofishing from the a) lower Murray River at Chowilla and b) the floodplain and c) gorge geomorphic regions in March/April 2014.

4.2 Lateral movements of fish

Environmental variables

Flow to South Australia is shown in Section 1.2 (Figure 2), whilst main channel discharge and water level at the locks nearest the wetland study sites are presented in Figure 16. Discharge regime at Locks 3 and 4 closely followed the discharge at the South Australian border. Water level within one weir pool closely followed discharge whilst the other weir pool had contrasting responses throughout the study period (Figure 16).

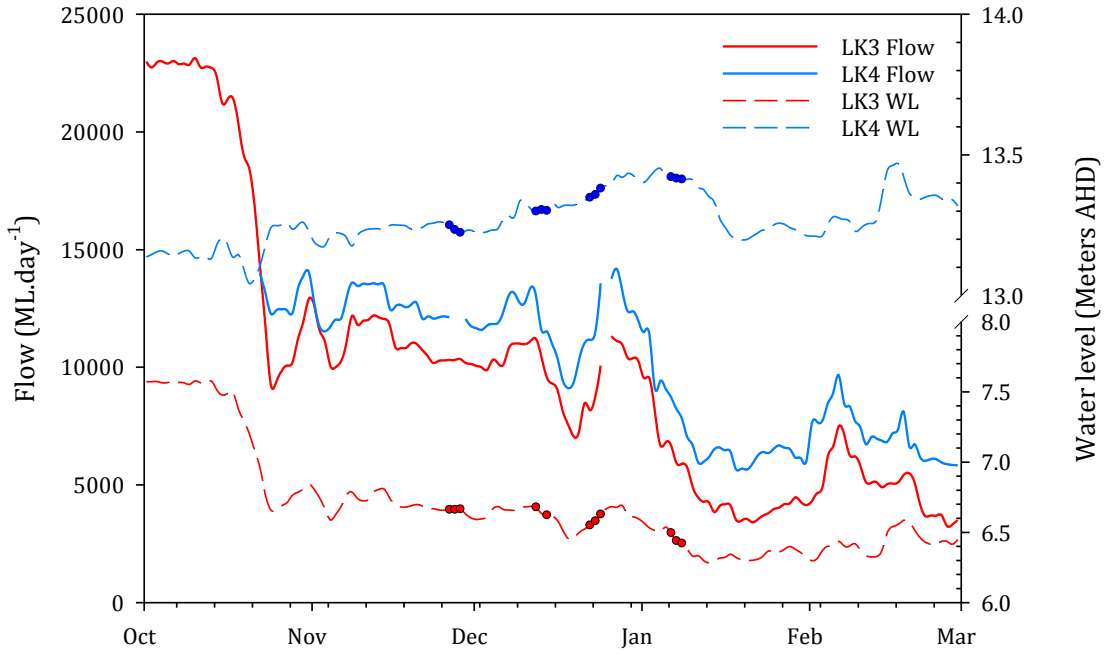


Figure 16. Discharge (solid lines) and water level (dashed lines) at Lock 3 (near Hart Lagoon) and Lock 4 (near Martin's Bend). Circles represent sampling events at each site (green: Hart Lagoon; blue: Martin's Bend).

Of 10 environmental variables from the studied wetlands, several were correlated (Spearman test, Table 7). For each pair of correlated variables, one was removed from subsequent analysis. Consequently the five environmental variables considered were transparency, dissolved oxygen, flow in the main channel, temperature and inlet flow.

Table 7. Spearman rank order correlation for environmental variables recorded at Hart Lagoon and Martin's Bend. Significant *P* values *: *P*<0.05; **: *P*<0.01. MC=main channel.

Hart Lagoon	Inlet flow	DO	Cond.	pH	Temp.	Turb.	Transp.	Flow (MC)	Water level (MC)
Inlet height	0.376	0.197	-0.718	0.387	-0.226	-0.219	0.446	-0.578	-0.413
P	*0.215*	*0.527*	*0.007***	*0.206*	*0.470*	*0.484*	*0.136*	*0.045**	*0.173*
Inlet flow		0.224	-0.347	0.175	0.427	0.357	0.308	-0.203	-0.119
P		*0.470*	*0.253*	*0.572*	*0.157*	*0.243*	*0.317*	*0.513*	*0.699*
DO			-0.060	0.622	0.308	0.147	-0.103	-0.259	-0.455
P			*0.834*	*0.029**	*0.317*	*0.635*	*0.733*	*0.402*	*0.130*
Conductivity				0.053	0.378	0.595	-0.830	-0.074	-0.182
P				*0.852*	*0.215*	*0.039**	*<0.001***	*0.800*	*0.557*
pH					0.266	0.308	-0.149	-0.799	-0.820
P					*0.389*	*0.317*	*0.635*	*<0.001***	*<0.001***
*Temperature						0.629	-0.251	-0.235	-0.249
P						*0.026**	*0.415*	*0.442*	*0.415*
Turbidity							-0.506	-0.319	-0.291
P							*0.089*	*0.295*	*0.340*
Transparency								0.209	0.372
P								*0.498*	*0.224*
Flow main channel									0.916
P									*<0.001***

Table 7 cont.

Martin's Bend	Inlet flow	DO	Cond.	pH	Temp.	Turb.	Transp.	Flow (MC)	Water level (MC)
Inlet height	-0.152	0.018	0.300	-0.519	0.032	0.307	-0.270	0.818	-0.540
P	*0.619*	*0.939*	*0.329*	*0.080*	*0.904*	*0.317*	*0.377*	*<0.001***	*0.066*
Inlet flow		0.441	-0.135	0.394	-0.070	-0.319	0.131	0.009	0.200
P		*0.143*	*0.667*	*0.197*	*0.817*	*0.295*	*0.667*	*0.974*	*0.513*
DO			-0.032	0.404	-0.466	-0.144	-0.171	0.158	-0.021
P			*0.904*	*0.181*	*0.117*	*0.635*	*0.588*	*0.603*	*0.939*
Conductivity				0.004	0.382	0.312	-0.736	0.046	0.326
P				*0.974*	*0.206*	*0.306*	*0.005***	*0.869*	*0.284*
pH					0.235	-0.543	0.116	-0.661	0.676
P					*0.442*	*0.062*	*0.699*	*0.017**	*0.014**
*Temperature						0.077	-0.172	-0.221	0.210
P						*0.800*	*0.572*	*0.470*	*0.498*
Turbidity							-0.773	0.508	-0.322
P							*0.002***	*0.084*	*0.295*
Transparency								-0.313	-0.081
P								*0.306*	*0.783*
Flow main channel									-0.785
P									*0.001***

Catch summary, species richness and abundance

A total of 29,023 fish from 15 species were sampled during this study. Throughout all sampling events, 2,535 fish from 11 fish species (eight native and three exotic) were collected in HL inlet and 13,833 fish from 15 species (11 native and four exotic) were sampled in MB inlet (Table 8). HL inlet was numerically dominated by the two natives carp gudgeon (*Hypseleotris* spp.) and bony herring (*Nematalosa erebi*), representing 40% and 36.8% of the total catch. In MB inlet, the exotic Gambusia (*Gambusia holbrooki*) was an order of magnitude more abundant than the two natives that followed in abundance, namely, carp gudgeon and bony herring, representing 69.5%, 12.8% and 9.9%, respectively. Four species were recorded on a single occasion, congolli at HL inlet and silver perch, oriental weatherloach (*Misgurnus anguillicaudatus*) and

freshwater catfish (*Tandanus tandanus*) at MB inlet. The number of species observed in each trip ranged between 7–10 at HL inlet and 8–11 at MB inlet (Table 8).

Within the wetlands, a total of 12,655 individuals from eight fish species were collected, with 94% of the catch coming from MB. The two native species, freshwater catfish and unspecked hardyhead (*Craterocephalus stercusmuscarum fulvus*) were observed on a single occasion in HL and MB, respectively. Concurrently, the three exotic species Gambusia, carp and goldfish (*Carassius auratus*) were present in both wetlands throughout all SE (Table 9). Total abundances in HL was dominated by carp (31.3%) followed by carp gudgeon (25.6%), bony herring (20.2%) and Gambusia (19.4%). Within MB the same pattern as the MB inlet was observed with Gambusia (85.2%) dominating; followed by the two native species carp gudgeon (5.96%) and bony herring (5.1%), whilst the exotic carp only represented 3.5% of the catch. Total CPUE within wetlands did not vary among sampling events (One-way ANOVA, HL: $F_{3,32}$=2.276, P=0.099; MB: $F_{3,32}$=0.837, P=0.484).

Table 8. Summary of mean (n = 9 net days trip^{-1}) catch-per-unit-effort (fish hour^{-1}) per species in the inlets of the two studied wetlands. IN and OUT represent fish caught moving "in to" and "out of" the wetlands.

	Hart Lagoon Inlet										Martin's Bend Inlet									
Trip	1		2		3		4		Total		1		2		3		4		Total	
Species\direction	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**	**IN**	**OUT**
Gambusia holbrooki	1.5	-	2.7	0.8	1.9	0.4	2.8	5.8	**8.9**	**7.0**	117.5	156.5	393.0	162.2	366.7	77.6	587.6	1686.4	**1464.8**	**2082.7**
Philypnodon grandiceps	0.8	0.8	-	0.3	1.8	1.4	2.6	4.6	**5.3**	**7.2**	13.4	11.9	20.2	11.6	17.7	6.5	35.0	35.7	**86.4**	**65.7**
Hypseleotris spp.	30.6	15.7	78.6	18.2	131.8	28.8	19.9	32.6	**260.9**	**95.3**	114.5	119.2	58.5	29.3	164.6	51.3	57.7	66.2	**395.2**	**266.0**
Cyprinus carpio	6.4	2.9	3.8	2.3	2.4	3.8	86.1	11.0	**98.6**	**20.0**	7.8	10.3	3.8	2.2	5.0	3.0	182.1	6.8	**198.7**	**22.2**
Carassius auratus	1.9	2.9	0.3	0.6	0.7	-	3.1	6.7	**6.0**	**10.2**	1.1	0.7	0.4	-	0.7	0.4	-	-	**2.2**	**1.1**
Nematalosa erebi	9.6	19.8	24.5	35.4	65.1	65.9	43.7	60.4	**142.9**	**181.5**	62.3	12.4	100.8	22.3	107.0	48.7	115.3	38.3	**385.4**	**121.7**
Macquaria ambigua ambigua	0.5	-	-	-	-	-	0.3	0.3	**0.8**	**0.3**	0.7	0.4	2.9	0.3	0.9	-	2.2	-	**6.6**	**0.7**
Retropinna semoni	1.0	11.9	-	2.7	-	-	1.0	3.3	**2.1**	**17.8**	-	-	1.1	-	0.8	-	-	-	**1.9**	**-**
Craterocephalus stercusmuscarum fulvus	0.3	1.7	0.3	-	-	-	0.7	2.1	**1.4**	**3.8**	3.1	-	-	-	1.2	-	0.7	-	**5.0**	**-**
Pseudaphritis urvillii	0.3	-	-	-	-	-	-	-	**0.3**	**-**	-	-	-	-	-	-	-	-	**-**	**-**
Philypnodon macrostomus	-	-	-	-	-	-	-	-	**-**	**-**	0.3	0.4	-	-	-	-	-	0.4	**0.3**	**0.8**
Melanotaenia fluviatilis	-	-	-	0.4	0.3	0.7	1.9	6.0	**2.2**	**7.2**	-	-	-	-	0.4	0.3	12.4	4.1	**12.8**	**4.4**
Bidyanus bidyanus	-	-	-	-	-	-	-	-	**-**	**-**	-	-	-	-	0.4	0.7	-	-	**0.4**	**0.7**
Misgurnus anguillicaudatus	-	-	-	-	-	-	-	-	**-**	**-**	-	-	-	-	-	-	0.4	-	**0.4**	**-**
Tandanus tandanus	-	-	-	-	-	-	-	-	**-**	**-**	-	-	-	-	-	-	0.4	-	**0.4**	**-**
Number of species	10		9		7		10				9		8		11		11			
Shannon's H	1.56		1.02		0.90		1.57				1.27		0.98		1.19		0.78			

Table 9. Summary of mean CPUE per trip (n = 9 net days trip^{-1}) per species in the two sampled wetlands.

	Hart Lagoon					Martin's Bend				
Species \ Trip	1	2	3	4	**Total**	1	2	3	4	**Total**
Gambusia holbrooki	0.69	5.37	6.96	3.96	**16.98**	236.96	259.95	390.08	400.58	**1287.57**
Phylopnodon grandiceps	-	0.25	-	0.23	**0.48**	-	0.23	-	0.12	**0.35**
Hypseleotris spp.	11.96	7.30	2.36	2.08	**23.71**	27.15	24.34	26.11	12.28	**89.88**
Cyprinus carpio	11.95	4.12	3.94	9.00	**29.01**	13.18	8.83	5.32	24.20	**51.54**
Carassius auratus	0.70	0.12	0.24	1.43	**2.49**	0.71	1.86	0.99	0.00	**3.56**
Nematalosa erebi	3.93	3.31	2.41	8.86	**18.52**	4.05	27.12	37.25	9.90	**78.32**
Craterocephalus stercusmuscarum fulvus	-	-	-	-	**0.00**	0.12	-	-	-	**0.12**
Tandanus tandanus	-	-	-	0.11	**0.11**	-	-	-	-	**0.00**
Total	**29.24**	**20.48**	**15.92**	**25.67**	**91.30**	**282.16**	**322.34**	**459.76**	**447.07**	**1511.34**

Lateral movements of fish

An overall analysis of fish assemblages moving through inlets at both wetlands revealed several significant interaction terms suggesting that no consistent pattern could be determined (Table 10). Therefore a broken down analysis of the factors was necessary.

Table 10. Three-way PERMANOVA for factors Site (HL and MB), Sampling events (SE1: stable, SE2: receding, SE3: rising EW, SE4: receding EW) and direction of movement (IN: into the wetland, OUT: out of the wetland). Data was 4th root transformed. Significant *P* values *: *P*<0.05; **: *P*<0.01. EW: environmental water.

Source	df	MS	Pseudo-F	*P*(perm)
Site (Si)	1	9454.00	13.989	0.002**
Sampling event (SE)	3	1573.80	6.861	0.001**
Direction of movement (DM)	1	707.68	2.352	0.100
Si x SE	3	675.84	2.946	0.006**
Si x DM	1	1623.40	6.990	0.008**
SE x DM	3	300.94	1.312	0.216
Si x SE x DM	3	232.25	1.012	0.440
Res	32	229.40		

The fish assemblages collected moving "IN" to wetlands were pooled and analysed in relation to the environmental variables selected (e.g. inlet flow, dissolved oxygen, temperature, transparency and flow in the main channel). DISTLM procedure revealed that temperature, inlet flow and flow in the main channel had a significant influence in fish assemblage structure moving "IN" (Table 11).

Table 11. Hierarchical classification of environmental variables that provide the best solution in explaining fish assemblages moving "IN" to Hart Lagoon and Martin's Bend inlets as revealed by DISTLM procedure. Significant *P* values *: *P*<0.05; **: *P*<0.01.

IN				
Variable	Adjusted R^2	Pseudo-*F*	*P*	Cumulative proportion
Temperature	0.1034	3.653	0.011*	0.142
Flow (main channel)	0.1739	2.876	0.035*	0.246
Inlet flow	0.2415	2.871	0.037*	0.340

The distance based redundancy analysis (dbRDA) of the assemblages moving "IN" to wetlands showed a clear separation along the x-axis between fish assemblages from each wetland (Figure 17). Those from HL were positively associated with increased inlet flows, whilst those from MB were positively associated with increased dissolved oxygen and decreased inlet flow. For fish assemblages from both wetlands the vertical distribution showed a separation between SE4 (descending EW) and all previous sampling events which had a positive association to increased flow in the main channel (Figure 17).

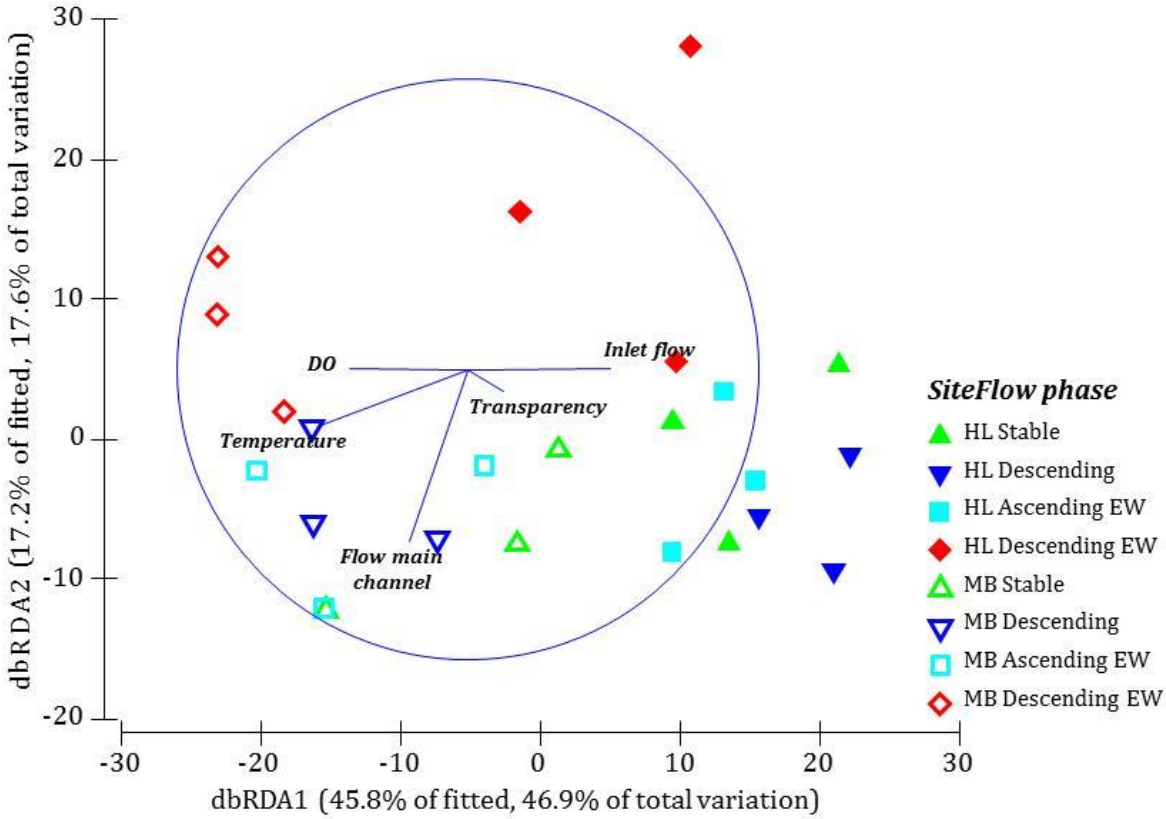


Figure 17. dbRDA ordination of the fitted model of fish assemblages moving "IN" through Hart Lagoon and Martin's Bend inlets versus the environmental predictor variables during four sampling events in the Lower Murray River in 2013-14.

When analysing fish assemblages collected moving "OUT' of wetlands in relation to environmental variables, the DISTLM procedure detected two variables with a significant influence, namely inlet flow and temperature (Table 12).

Table 12. Hierarchical classification of environmental variables that provide the best solution in explaining fish assemblages moving "OUT" of Hart Lagoon and Martin's Bend inlets as revealed by DISTLM procedure. Significant *P* values *: *P*<0.05; **: *P*<0.01.

OUT				
Variable	Adjusted R^2	Pseudo-*F*	*P*	Cumulative proportion
Inlet flow	0.1196	4.123	0.007**	0.158
Temperature	0.2120	3.581	0.018*	0.281

The dbRDA of fish assemblages moving "OUT" of HL and MB inlets revealed a distinct separation of wetlands along the x-axis as previously observed for fish moving "IN" with environmental variables having similar associations with fish assemblages from each wetland (Figure 18).

Patterns of fish movement between the main river channel and wetlands were complex and highly variable among flow phases at each wetland, and such differences were not consistent among wetlands (Table 9; Figure 17; Figure 18). As such, fish movement patterns for each wetland were analysed and presented separately.

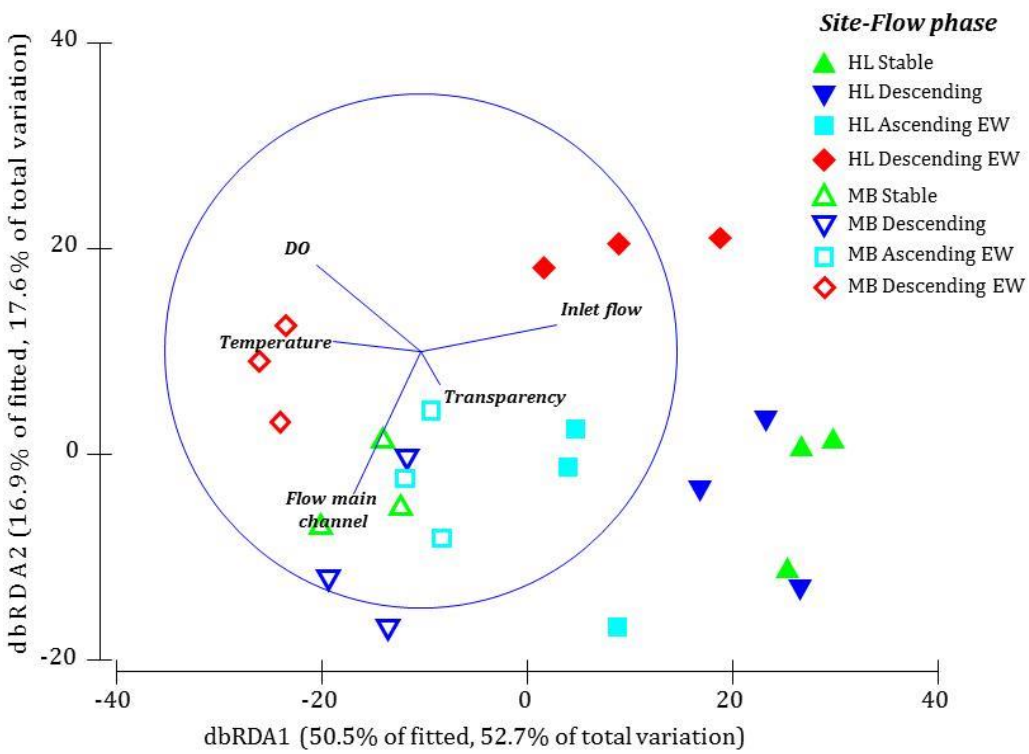


Figure 18. dbRDA ordination of the fitted model of fish assemblages moving "OUT" through Hart Lagoon and Martin's Bend inlets versus the environmental predictor variables during four sampling events in the Lower Murray River in 2013-14.

Hart Lagoon

Total CPUE in the HL inlet differed among sampling events (SE) (*P*=0.007), but not in direction of movement (DM) (*P*=0.051). There was a consistent increase in total CPUE as the sampling events progressed and Student-Newman-Keuls (SNK) pair-wise comparison showed that SE2 = SE1, SE3 and SE4, whilst SE1 < SE3 and SE4 and finally SE3 = SE4.

Fish movement in and out of HL was dominated by the native species carp gudegon and bony herring for the first three sampling events (i.e. SE1, SE2 and SE3). In the last sampling event (i.e. SE4), a 40 fold increase in carp abundances led to a dominance by this exotic species, which was followed, by the two natives carp gudegon and bony herring. Results of two-way (SE x DM) ANOVAs for each species that met assumptions are presented in Table 13.

Table 13. Results for 2-way ANOVAs of CPUE for *Gambusia holbrooki*, *Philypnodon gradiceps*, *Hypseleotris* spp., *Cyprinus carpio*, *Carassius auratus*, *Nematalosa erebi* and *Melanotaenia fluviatilis* in Hart Lagoon. *Retropinna semoni, Craterocephalus stercusmuscarum fulvus* and *Macquaria ambigua ambigua* were present but did not meet assumptions and were excluded from analysis. (NS: not significant; *: *P*<0.05; **: *P*<0.01).

Hart Lagoon	*G. holbrooki*	*P. grandiceps*	*Hyp spp.*	*C. carpio*	*C. auratus*	*N. erebi*	*M. fluviatilis*
Direction of movement	NS	NS	*	NS	NS	NS	NS
Sampling event	NS	**	NS	**	NS	**	**
DM x SE	NS	NS	NS	NS	NS	NS	NS

No differences for CPUE in DM or among SE were detected for the exotic species Gambusia; however, carp showed a significant increase in CPUE (Figure 19) in the last sampling event mainly driven by YOY fish (Figure 20), whereas no differences were detected for DM.

The native species for which ANOVA detected a significant difference in CPUE among SE were flathead gudgeon (*Philypnodon grandiceps*), bony herring and Murray rainbowfish (*Melanotaenia fluviatilis*), whilst differences in DM were detected for carp gudgeon (Table 13). Pair-wise comparison (SNK) among SE for flathead gudgeon showed: SE1 = SE3 and SE4, whilst SE2 = SE1 and SE3 and finally SE3 = SE4; for bony herring showed: SE2 = SE1, SE3 and SE4, whilst SE1 < SE3 and SE4 and finally SE3 = SE4;

and for Murray rainbowfish showed: SE1 = SE2 and SE3, SE4 > SE1, SE2 and SE3. Comparison (SNK) of CPUE of carp gudgeon in DM showed: IN > OUT (Figure 21).

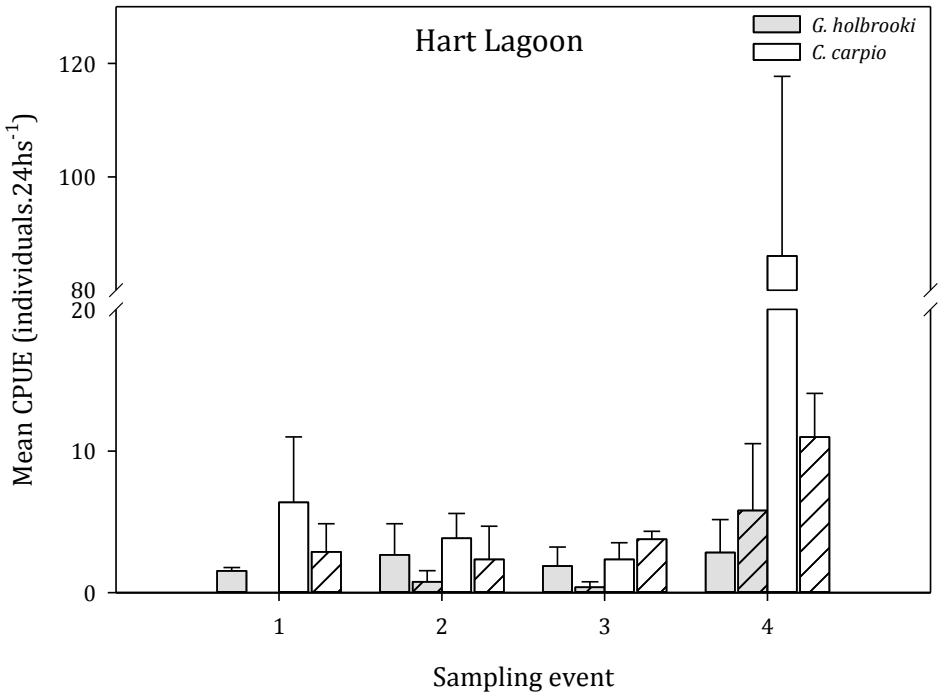


Figure 19. Mean (±S.E.) CPUE of exotic species collected moving IN (no pattern) and OUT (diagonal lines) of Hart Lagoon throughout the four sampling events.

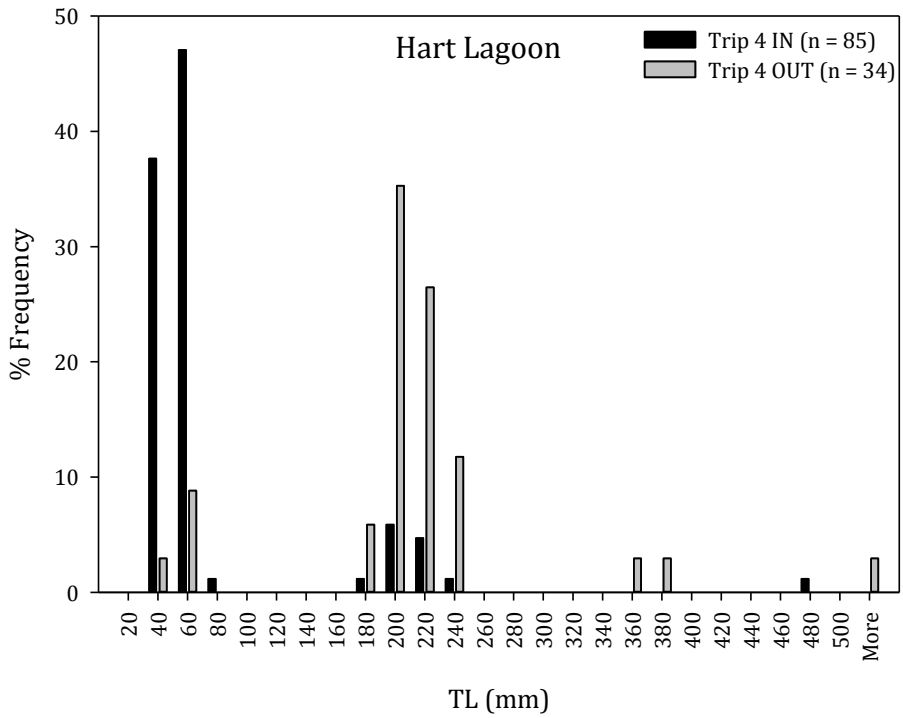


Figure 20. Size-frequency distribution of carp (*Cyprinus Carpio*) moving through the Hart Lagoon inlet.

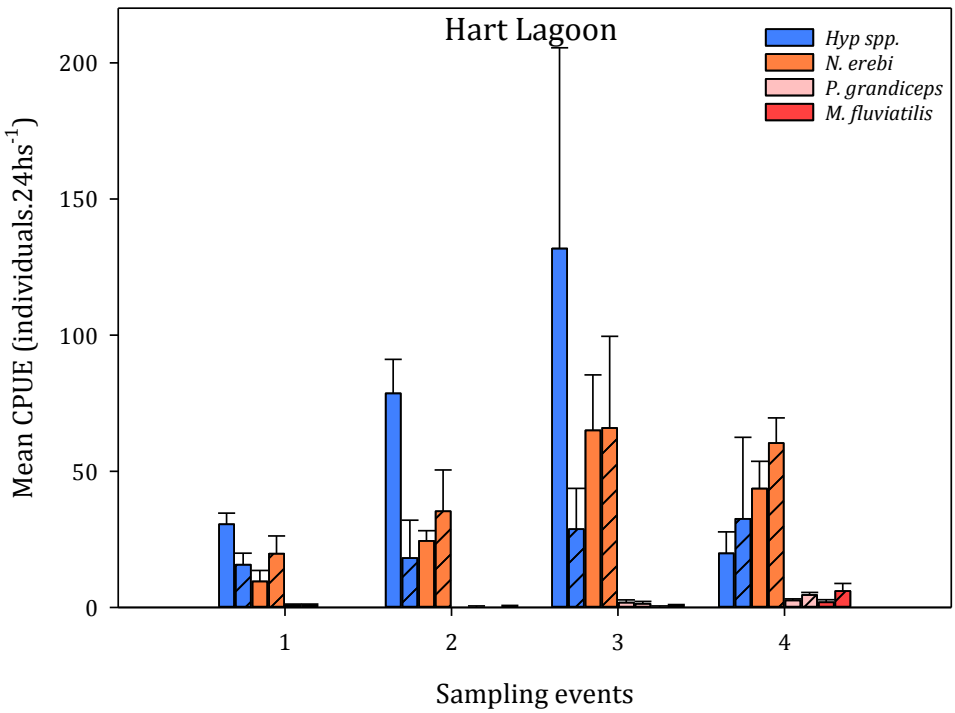


Figure 21. Mean (±S.E.) CPUE of native species collected moving IN (no pattern) and OUT (diagonal line) of Hart Lagoon throughout the four sampling events.

Within the wetland, CPUE of Gambusia showed significant differences among SE (*P*=0.031), with SNK pair-wise comparison only detected differences between SE1 < SE3. Mean CPUE of carp within HL also showed significant differences among SE (*P*=0.001). Pair-wise comparison showed that SE1 > SE2 and SE3, with no differences detected among all other comparisons (Figure 22). CPUE of goldfish did not meet normality assumptions and was not analysed.

Among the native fish species present in HL, bony herring was the only species that met ANOVA assumptions. ANOVA detected significant differences in SE (*P*=0.005). Pair-wise comparison showed that SE4 > SE1, SE2, SE3 and no differences among other SE (Figure 22).

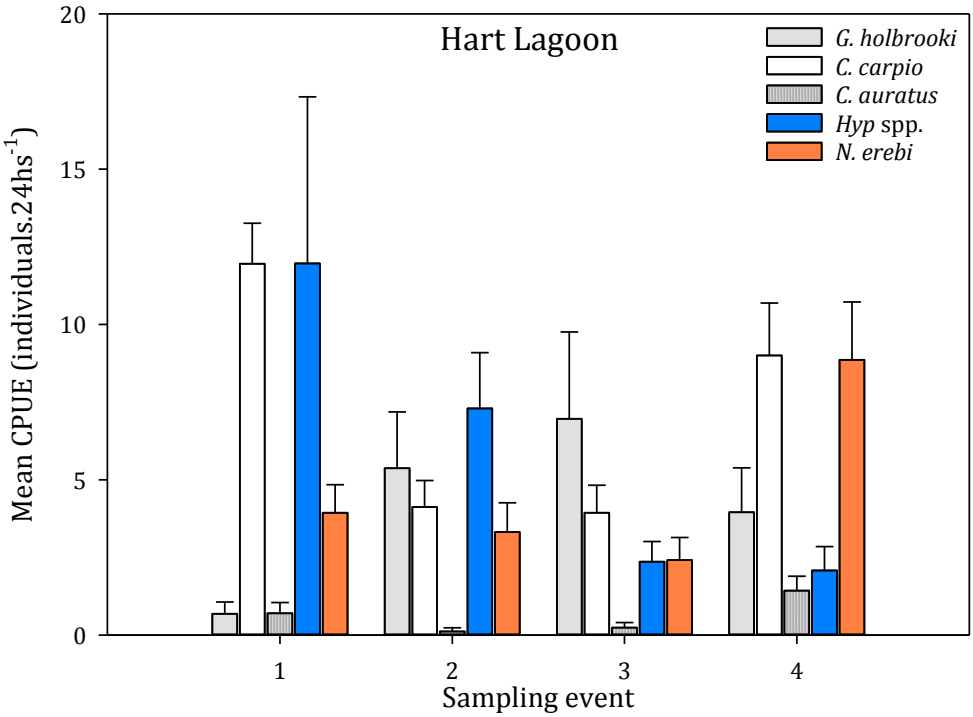


Figure 22. Mean (±S.E.) CPUE of exotic and abundant native species collected within Hart Lagoon wetland throughout the four sampling events.

Of the five environmental predictor variables available to investigate variability among fish assemblages in HL inlet (DistLM), flow in the main channel and transparency were identified as significant predictors (Figure 23; Table 14). There was a positive association between increased flow in the main channel and the stable (SE1), descending (SE2) and ascending EW (SE3) and a negative association with descending EW (SE4). Together flow in the main channel and transparency provided the best correlation between any combination of environmental variables and fish assemblages moving through HL inlet (BEST: correlation 0.314).

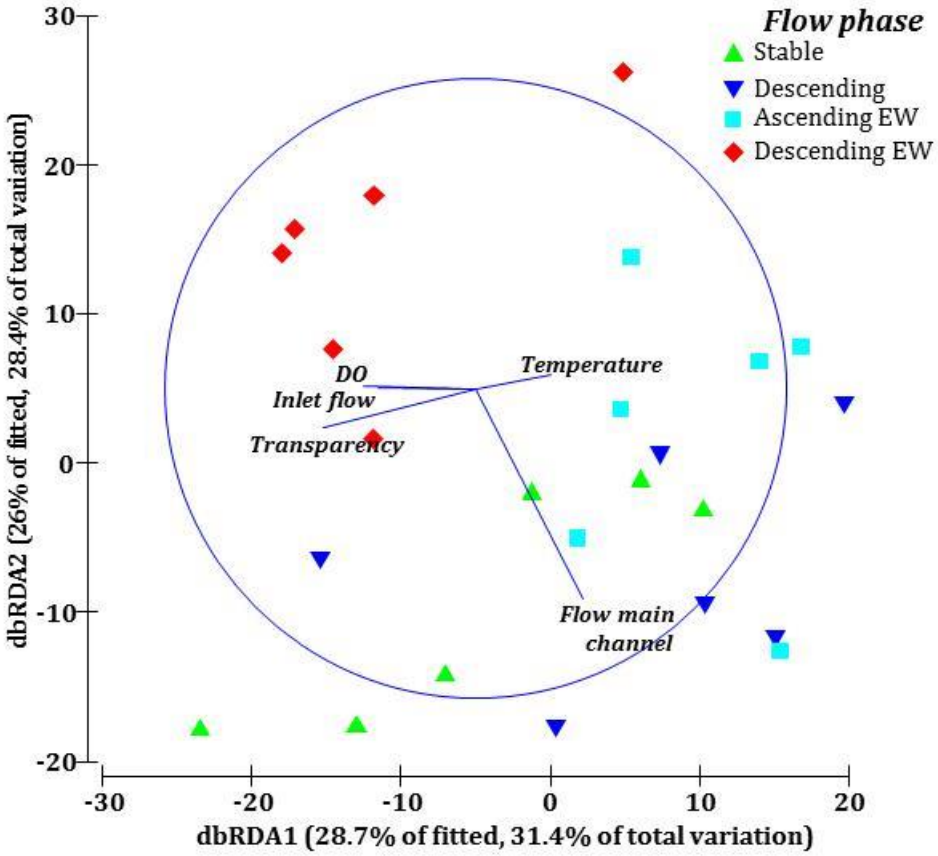


Figure 23. dbRDA ordination of the fitted model of fish assemblages moving through Hart Lagoon inlet versus the environmental predictor variables during four sampling events in the Lower Murray River in 2013-14.

Table 14. Hierarchical classification of environmental variables that provide the best solution in explaining fish assemblages moving through Hart Lagoon inlet as revealed by DISTLM procedure. Significant *P* values *: *P*<0.05; **: *P*<0.01.

Hart Lagoon				
Variable	Adjusted R^2	Pseudo-*F*	*P*	Cumulative proportion
Flow (main channel)	0.1553	5.228	0.001**	0.192
Transparency	0.2569	4.008	0.003**	0.322
Dissolved oxygen	0.2695	1.361	0.275	0.365

Martin's Bend

Total fish abundance (CPUE) moving IN and OUT of MB wetland showed a significant interaction term (*P*=0.002) between sampling events (SE) and direction of movement (DM), pair-wise comparisons are shown in Table 15. Gambusia represented 69.2 % of the total CPUE in MB with potential for bias of the overall CPUE. Therefore, ANOVA was

re-run excluding Gambusia, in order to identify potential patterns that may have been obscured by including Gambusia (Table 16).

Table 15. SNK pair-wise comparison of a) DM among SE and b) SE among DM. Significant *P* values *: *P*<0.05; **: *P*<0.01.

a)

SE	SNK	*P*
1	IN = OUT	0.890
2	IN > OUT	0.011*
3	IN > OUT	0.002**
4	IN < OUT	0.024*

b)

DM	SNK	*P*	DM	SNK	*P*
IN	SE1 = SE2	0.080	OUT	SE1 = SE2	0.399
	SE1 = SE3	0.085		SE1 = SE3	0.423
	SE1 < SE4	0.007**		SE1 < SE4	<0.001**
	SE2 = SE3	0.675		SE2 = SE3	0.681
	SE2 = SE4	0.144		SE2 < SE4	<0.001**
	SE3 = SE4	0.135		SE3 < SE4	<0.001**

Table 16. a) Results for ANOVA and b) SNK pair-wise comparison of total fish CPUE excluding Gambusia. Significant *P* values *: *P*<0.05; **: *P*<0.01.

a)

Source of variation	df	MS	F	*P*
Direction of movement	1	3.242	21.781	<0.001**
Sampling event	3	0.494	3.316	0.047*
DM x SE	3	0.165	1.109	0.374
Residual	16	0.149		

b)

	SNK	*P*
Direction of movement	IN > OUT	<0.001**
Sampling event	SE1 = SE2	0.091
	SE1 = SE3	0.967
	SE1 = SE4	0.397
	SE2 = SE3	0.189
	SE2 < SE4	0.030*
	SE3 = SE4	0.214

Abundance data of four species moving in and out of MB wetland were analysed through ANOVA, namely Gambusia, carp gudgeon, bony herring and flathead gudgeon. Table 17 summarises ANOVA results for each species.

Table 17. Results for 2-way ANOVAs of CPUE for fish species in Martin's Bend. (NS: not significant; *; *P*<0.05; **; *P*<0.01).

Martin's Bend	*Gambusia holbrooki*	*Philypnodon grandiceps*	*Hypseleotris spp.*	*Nematalosa erebi*
Direction of movement	-	NS	*	**
Sampling event	-	**	**	NS
DM x SE	**	NS	NS	NS

The exotic fish Gambusia was the numerically dominant species within MB and ANOVA detected a significant interaction which reflects the differences observed in Figure 24, revealing the high level of variability in the abundances of fish moving in and out of the wetland among sampled events.

ANOVA detected significant differences in abundances (CPUE) of the small-bodied native flathead gudgeon among SE (SE4 > SE1 = SE2 = SE3) and for carp gudgeon among both SE (SE1 > SE2 = SE4, SE1 = SE3, SE3 > SE2, SE3 = SE4) and DM (IN > OUT) (Table 17). Abundances (CPUE) of bony herring varied in DM (IN > OUT) but not throughout SE (Figure 25).

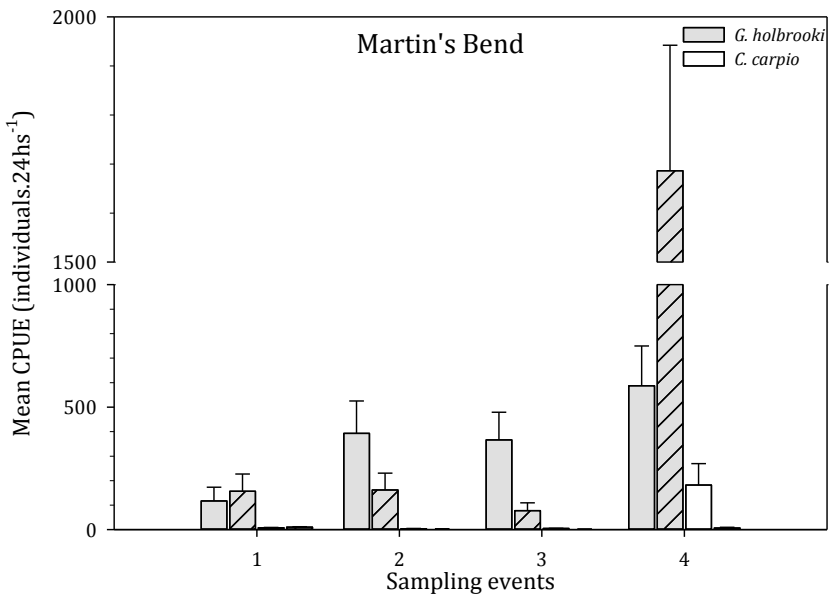


Figure 24. Mean (±S.E.) CPUE of exotic species collected moving IN (no pattern) and OUT (diagonal lines) of Martin's Bend throughout the four sampling events.

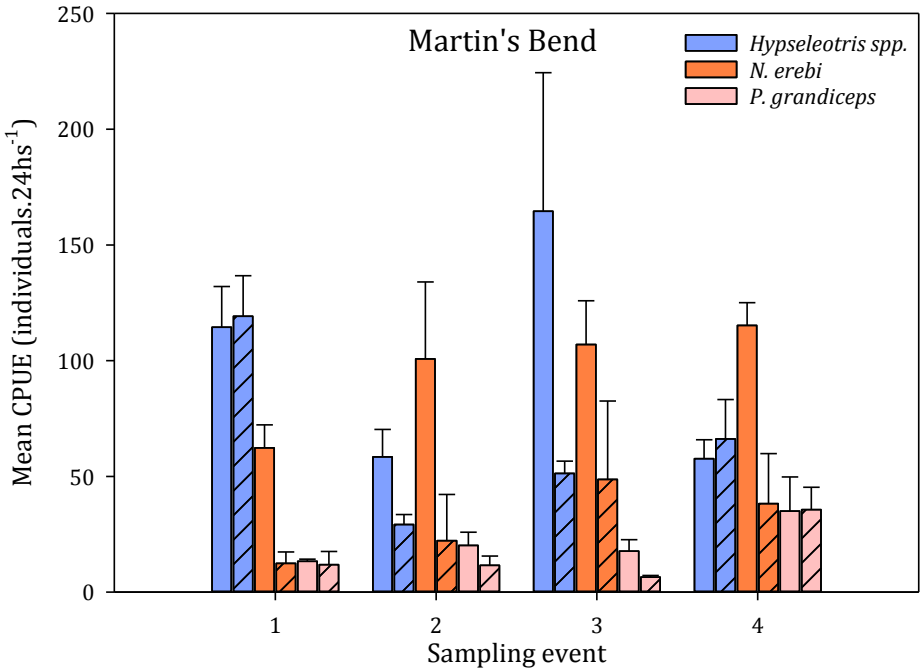


Figure 25. Mean (±S.E.) CPUE of abundant native species collected moving IN (full bar) and OUT (line) of Martin's Bend throughout the four sampling events.

The native large-bodied golden perch was recorded in all SE at MB inlet in relatively low abundance. Whilst no ANOVA was conducted, there was a clear trend towards greater abundances entering the wetland than those exiting (Figure 26). The size range of golden perch sampled in MB inlet was 20–234 mm, with two distinct size classes (20–30 mm: 52% and > 100 mm: 48%). Silver perch was the other native large-bodied species encountered in MB inlet, albeit only in SE3.

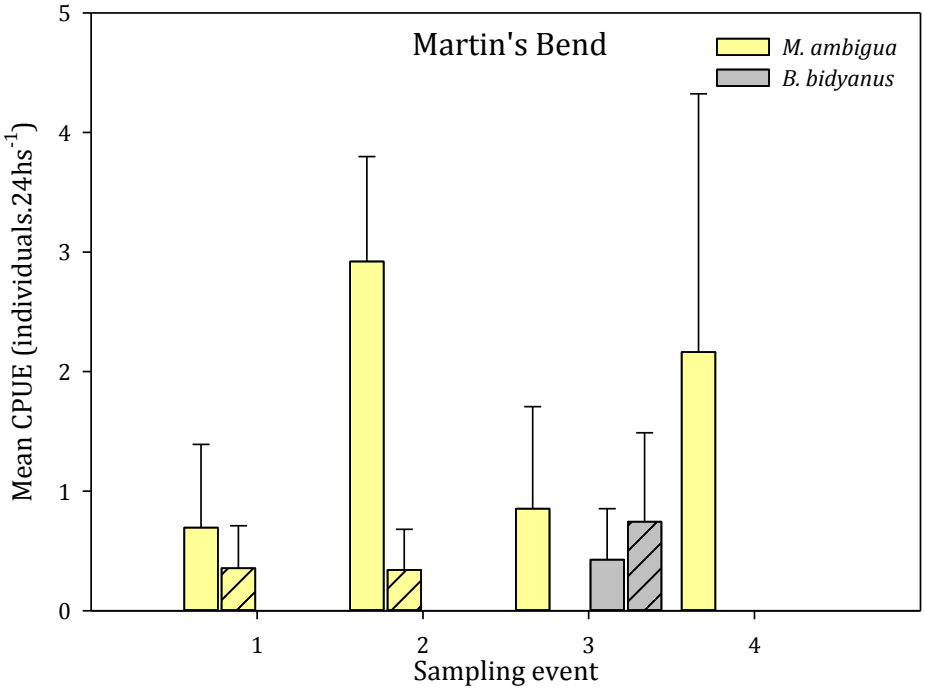


Figure 26. Mean (±S.E.) CPUE of native flow-cued spawning species collected moving IN (full bar) and OUT (line) of Martin's Bend throughout the four sampling events.

Within the wetland, no differences in CPUE of Gambusia were detected among SE (*P*=0.474), whilst differences in abundances of carp were detected (*P*=0.034) with SNK pair-wise comparison showing SE4 > SE3 and no differences in all other comparisons (Figure 27). CPUE of goldfish did not meet normality assumptions and was not analysed.

Among the native fish species present in MB, ANOVA detected significant differences in bony herring abundances among SE (*P*<0.001). Pair-wise comparison showed that SE3 = SE2 > SE4 = SE1 (Figure 27). No differences were detected in CPUE of carp gudgeon (*P*=0.195).

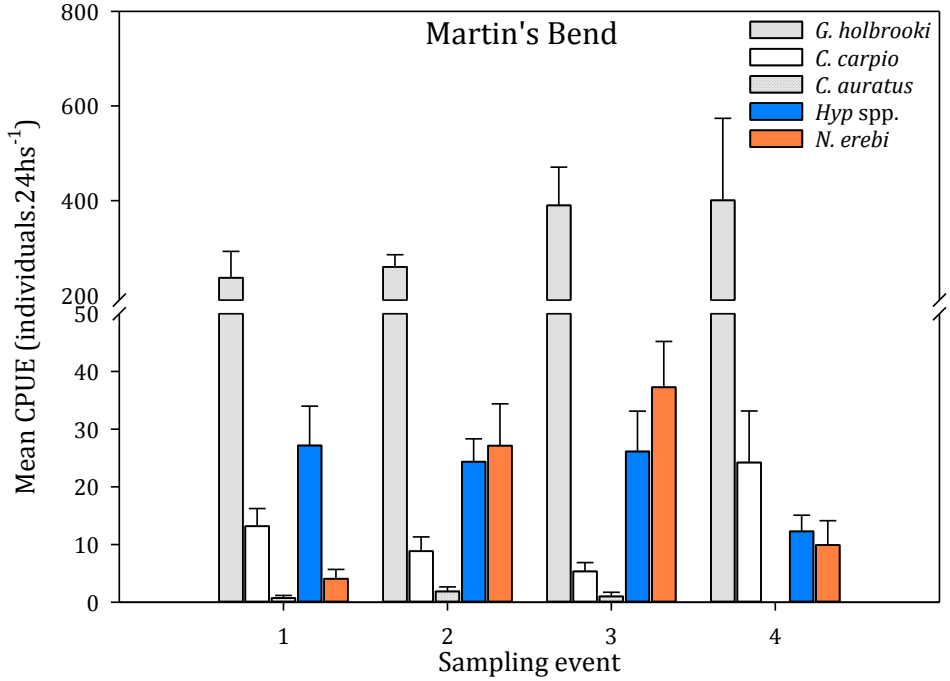


Figure 27. Mean (±S.E.) CPUE of exotic and abundant native species collected within Martin's Bend wetland throughout the four sampling events.

Of the five environmental predictor variables analysed by DistLM for fish assemblages in MB inlet, flow in the main channel was the only one identified as a significant predictor (Figure 28; Table 18). There was a positive association between increased flow in the main channel and the stable (SE 1), descending (SE 2) and ascending EW (SE 3) and a negative association with descending EW (SE 4). Flow in the main channel alone obtained the best correlation between any combination of environmental variables and fish assemblages moving through MB inlet (BEST: correlation 0.433).

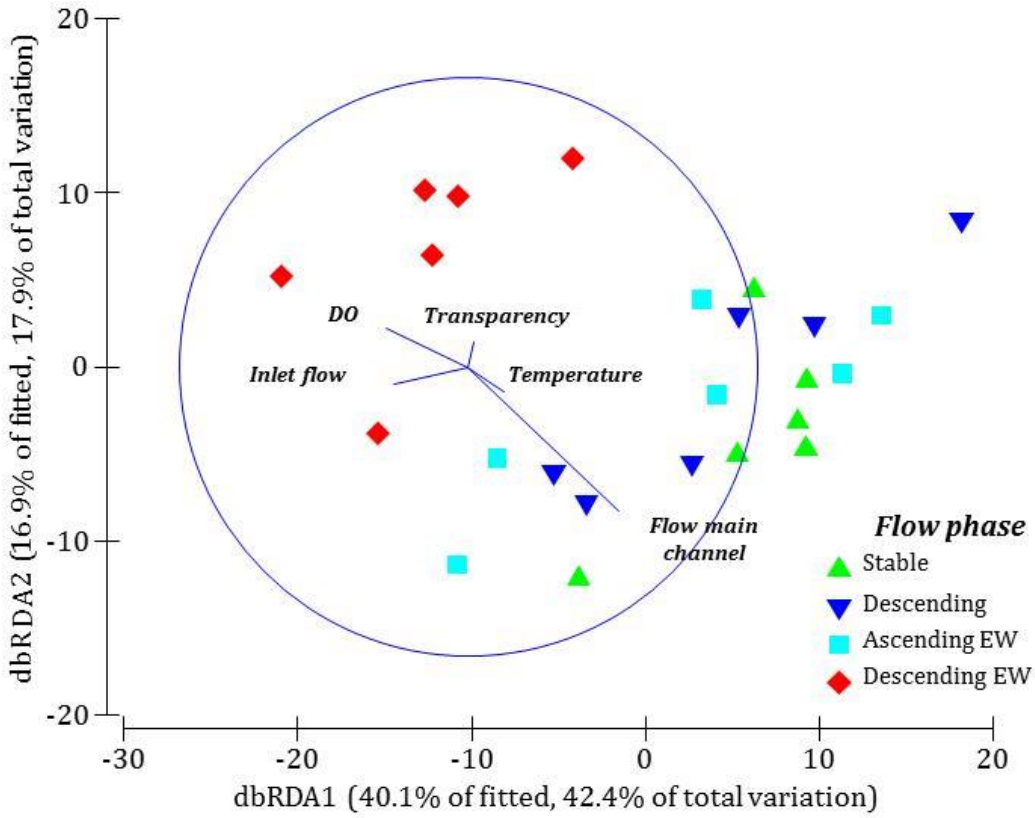


Figure 28. dbRDA ordination of the fitted model of fish assemblages moving through Martin's Bend inlet versus the environmental predictor variables during four sampling events in the Lower Murray River in 2013-14.

Table 18. Hierarchical classification of environmental variables that provide the best solution in explaining fish assemblages moving through Martin's Bend inlet as revealed by DISTLM procedure. Significant *P* values *: *P*<0.05; **: *P*<0.01.

	Martin's Bend			
Variable	Adjusted R^2	Pseudo-*F*	*P*	Cumulative proportion
Flow (main channel)	0.1926	6.487	0.001**	0.228
Dissolved oxygen	0.2299	2.065	0.079	0.297

4.3 Dissolved and particulate material transport

The modelling suggests that environmental watering had no effect on salinity in the Lower Murray River (Figure 29). Similarly, there was only a minor effect within Lake Alexandrina with average salinities over 2013-2014 of 0.27 PSU with both sources of environmental water compared to 0.30 PSU without environmental water. However, within the Murray Mouth, salinity was reduced significantly as a result of environmental water delivery with average salinities over 2013-2014 of 21.42 PSU with both sources of

environmental water compared to 24.57 PSU without. This is equivalent to a 13% reduction. Furthermore, at the Murray Mouth electrical conductivity was up to 38.6 PSU lower with environmental water delivery at times (equivalent to a reduction of up to approximately 75%) as a result of increased inflows of lower salinity water into the Northern Coorong.

Based on the modelling outputs, environmental watering increased salt exports from the Lower Murray River, Lower Lakes and Coorong (Figure 30). For the Lower Murray River, exports increased steadily, from July 2013 to July 2014, initially associated with background flows but with environmental water thereafter. There was a total modelled export of approximately 462,000 tonnes from the Lower Murray River to the Lower Lakes during the study period, of which environmental water use contributed approximately 81,000 tonnes. This is equivalent to a 21% increase associated with environmental water. Environmental watering also increased exports from the Lower Lakes, although this was a smaller proportion (7% increase) than from the Lower Murray River. This was likely due to the effect of environmental water lowering salinity concentrations within the Lower Lakes. The cumulative exports from the Coorong peaked higher in early 2014 due to the higher flows and fell away thereafter as flows decreased. Overall there was an export of 831,000 tonnes of salt from the Coorong during 2013-2014 which was higher than from the Lower Lakes (Figure 30) due to higher salinities within the Northern Coorong than the Lower Lakes (Figure 29). The modelling also suggests that whilst the individual environmental water sources increased cumulative exports by 5-9% when provided in independently, when combined the environmental water sources they increased cumulated exports by 24%.

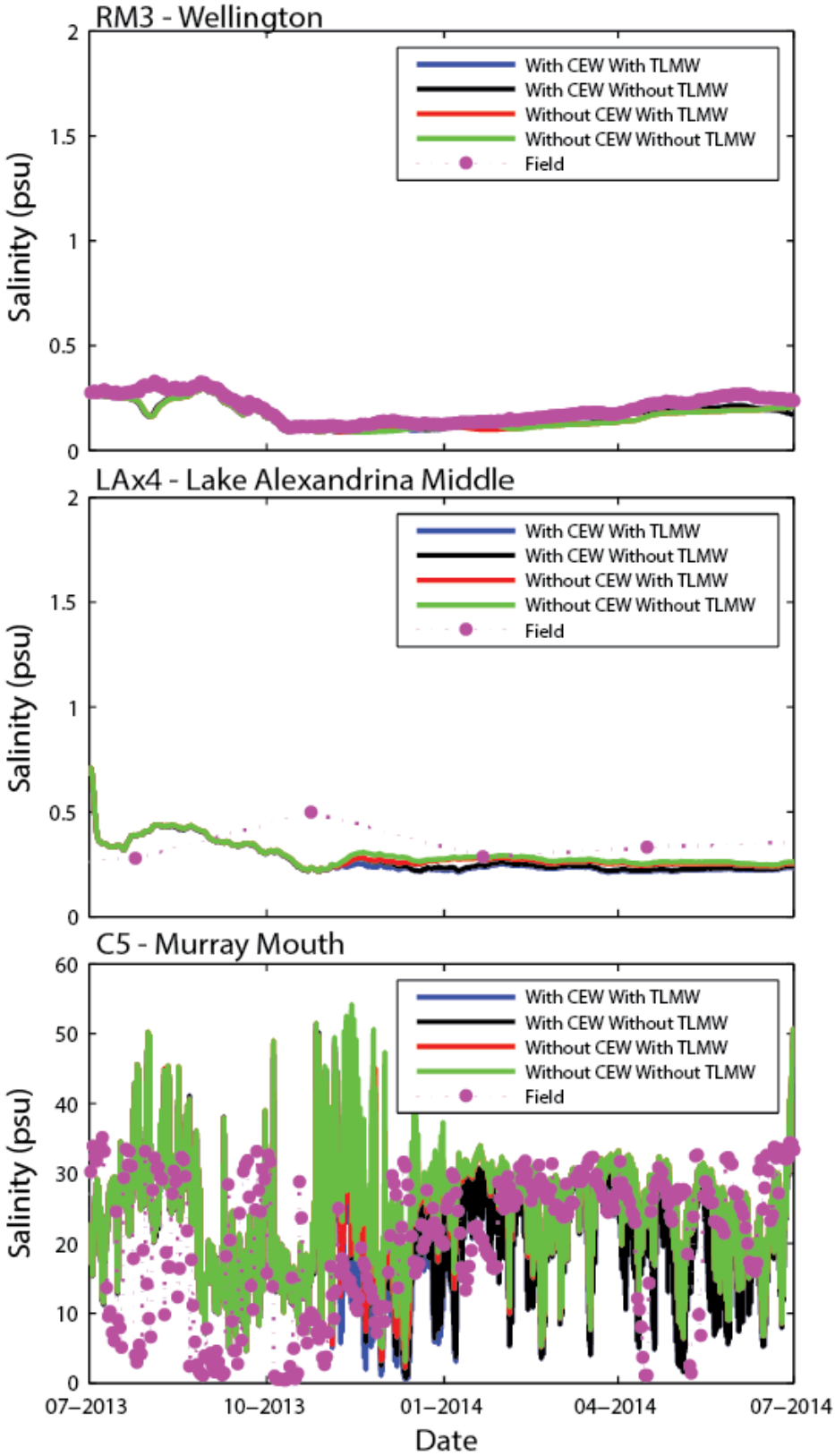


Figure 29. Observed and modelled practical salinity units (PSU) at selected sites. Scenarios include with and without both commonwealth environmental water (CEW) and The Living Murray water (TLMW).

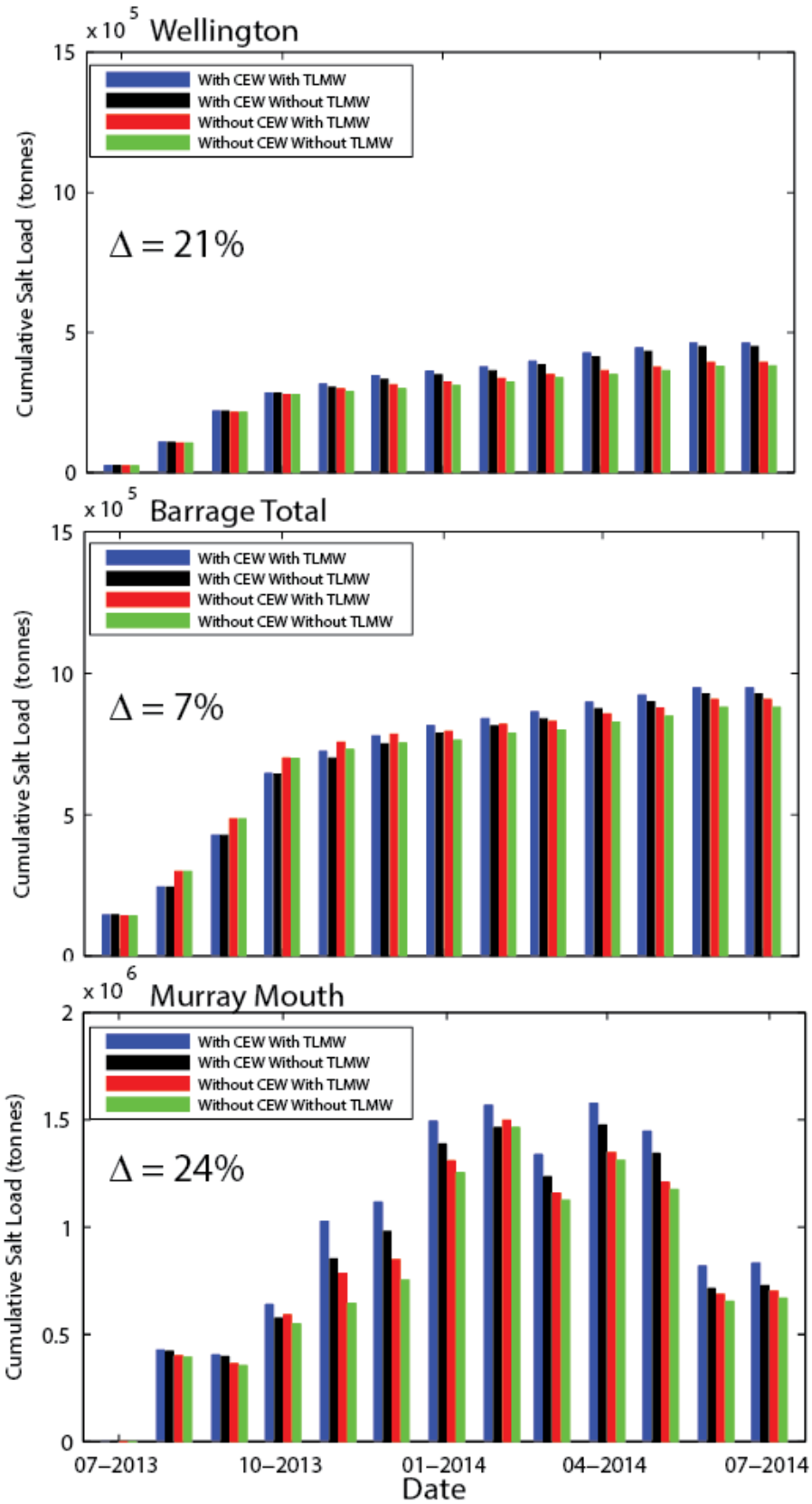


Figure 30. Modelled cumulative salt exports (net) with and without environmental water delivery. Scenarios include with and without both commonwealth environmental water (CEW) and The Living Murray water (TLMW). Δ is the percentage increase in loads associated with environmental water provisions (CEW and TLMW). Note the scale difference.

There were small differences in dissolved nutrient concentrations between the modelled scenarios, given uncertainties associated with the modelled outputs (Figure 31). The most apparent differences were higher phosphate concentrations and lower silica concentrations within Lake Alexandrina with environmental water; and lower ammonia concentrations in the Murray Mouth with environmental water.

Since there were only small differences in concentrations within the Lower Murray River, differences in modelled exports between the scenarios were largely a result of differences in discharge (Figure 32). Environmental water provisions increased exports of phosphate and silica from the Lower Lakes, although the overall export of phosphate was lower than that from the Lower Murray River due to retention within the Lower Lakes. There was little difference in the exports of ammonia from the Lower Lakes and Murray Mouth between scenarios due to opposing effects of increased flows, but at lower concentrations (Figure 31). Overall environmental water provisions increase ammonia exports from the Lower Murray River by approximately 20%, but there was negligible effect on exports from the Lower Lakes and Coorong. For phosphate, environmental water provisions increased the total exports from the Lower Murray River, Lower Lakes and Murray Mouth by 33%, 44% and 45%, respectively. Furthermore, the combined influence of both water sources was greater than that of the sum of individual water sources. For silica, environmental water increased the total exports from the Lower Murray River, Lower Lakes and Murray Mouth by approximately 13%, 21% and 21% of total exports, respectively.

There were only small differences in the modelled particulate nutrient concentrations with and without environmental water, particularly given uncertainties associated with the modelled outputs (Figure 33). Within the Lower Murray River, Lower Lakes and Coorong, particulate organic nitrogen and phosphorus exports were also higher with environmental watering and increased proportionally with discharge (Figure 34). Exports of particulate nutrients decreased from the Lower Murray River to the Murray Mouth, suggesting retention within the Lower Lakes and Coorong (Figure 34). Over the study period, the environmental watering increased total exports of particulate nutrients from the Murray River, Lower Lakes and Murray Mouth by approximately 20%, 30% and 35%, respectively.

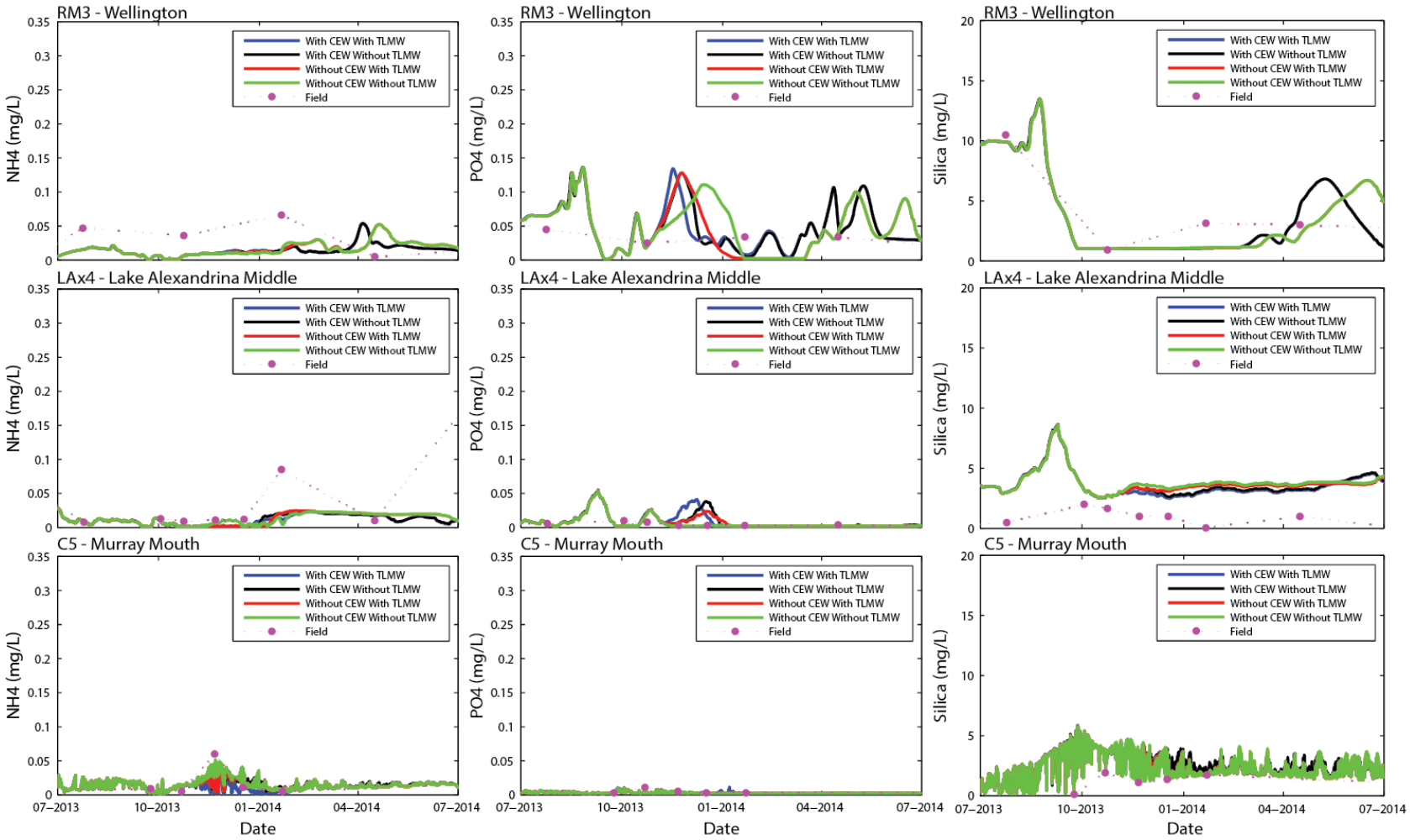


Figure 31. Observed and modelled ammonium (NH_4), phosphate (PO_4) and silica (SiO_2) concentrations at selected sites. Scenarios include with and without both commonwealth environmental water (CEW) and The Living Murray water (TLMW).

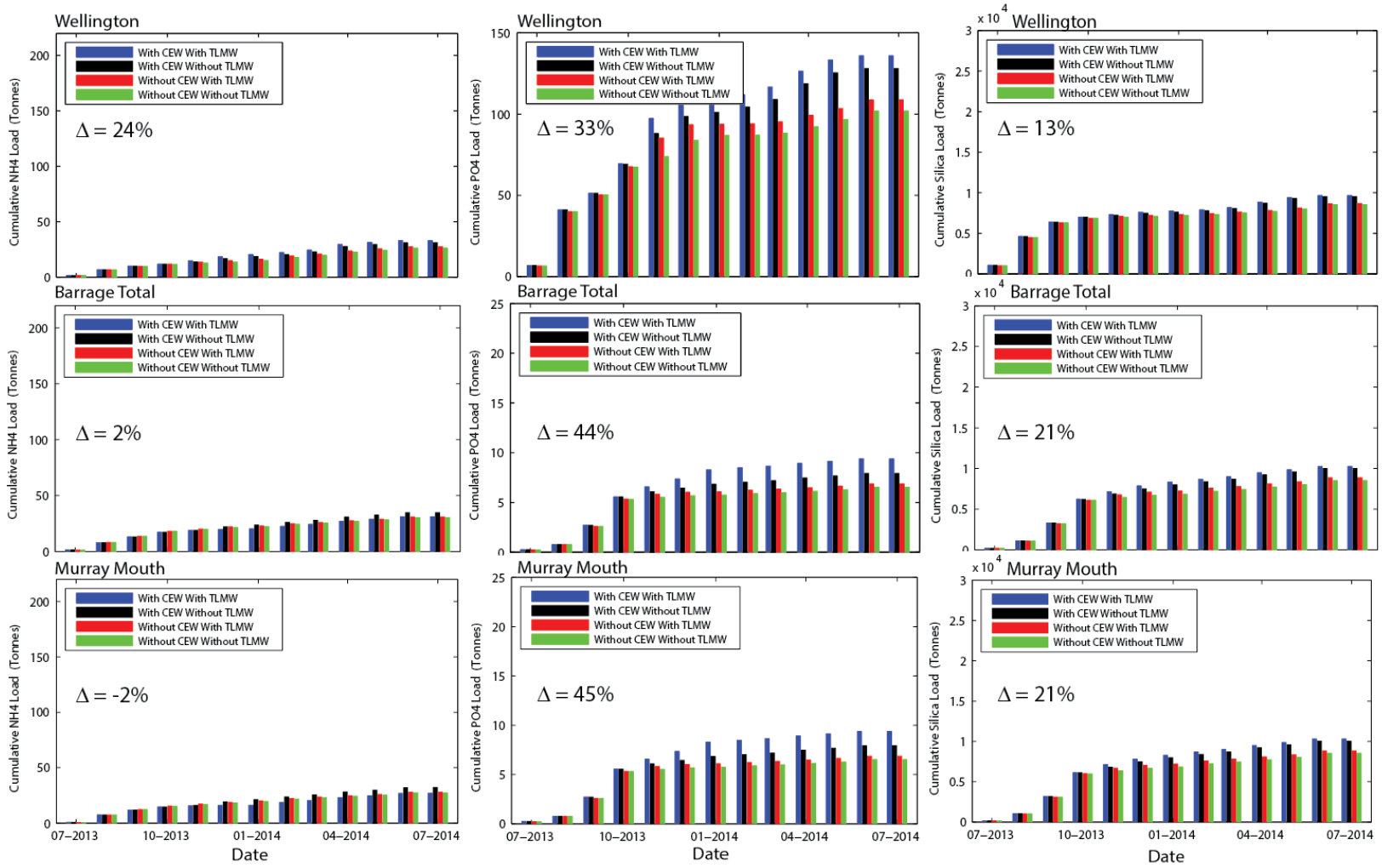


Figure 32. Modelled cumulative ammonium (NH_4), phosphate (PO_4) and silica (SiO_2) exports (net) with and without environmental water delivery. Scenarios include with and without both commonwealth environmental water (CEW) and The Living Murray water (TLMW). Δ Positive loads are exports passed a designated boundary, including Wellington (from the Murray River).

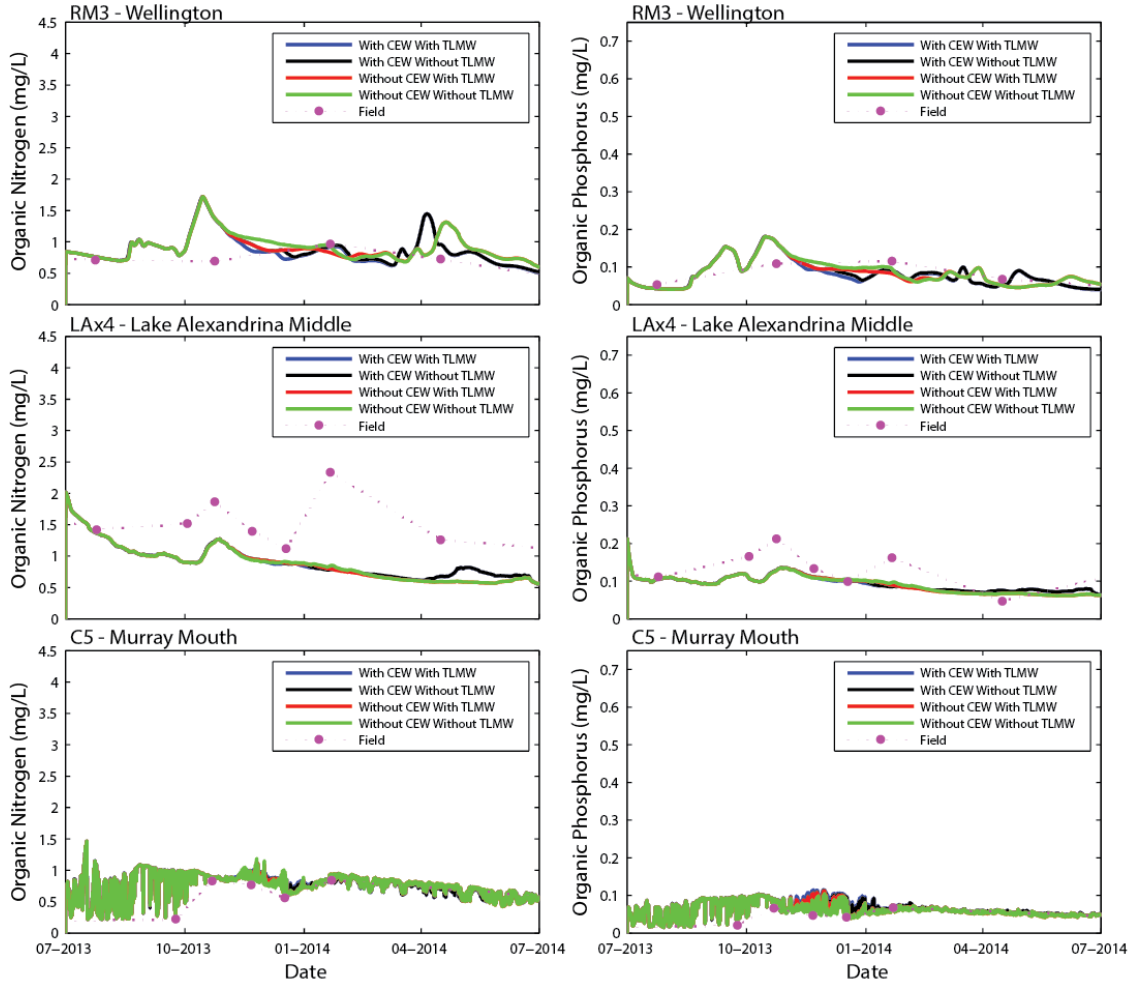


Figure 33. Observed and modelled particulate organic phosphorus concentrations at selected sites. Scenarios include with and without commonwealth environmental water (CEW) and The Living Murray water (TLMW).

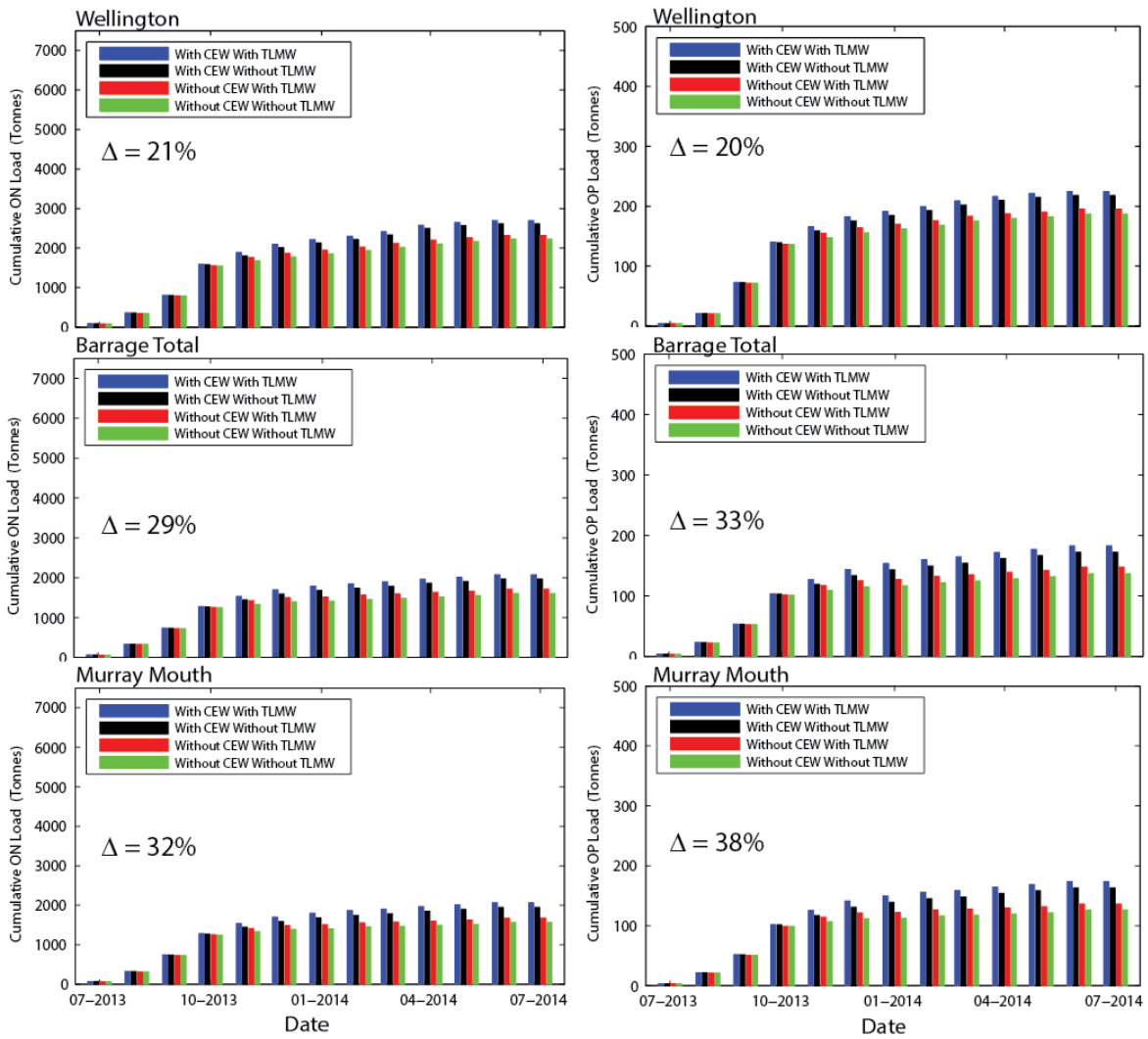


Figure 34. Modelled cumulative particulate organic nitrogen (ON) and phosphorus (OP) exports (net) with and without environmental water delivery. Scenarios include with and without both commonwealth environmental water (CEW) and The Living Murray water (TLMW). Δ is the percentage increase in loads associated with environmental water provisions (CEW and TLMW).

Given uncertainty in the modelled outputs of chlorophyll *a* concentrations, there was no apparent influence of environmental watering on chlorophyll *a* concentrations (Figure 35). As a result, differences in modelled exports reflected those of discharge with the additional environmental water provisions resulting in additional export of both chlorophyll *a* and total suspended solids (Figure 36). Overall, environmental water increased the total exports of phytoplankton biomass from the Lower Murray River, Lower Lakes and Murray Mouth by approximately 14%, 30% and 32%, respectively. Similarly, for total suspended solids environmental water increased the total exports from the Lower Murray River, Lower Lakes and Murray Mouth by approximately 27%, 23% and 40% respectively. However, unlike phytoplankton biomass there were greater exports from the Lower Murray River than from the Lower Lakes and Coorong although this was largely an artefact of the model underestimating turbidity levels within the Lower Lakes (Figure 35).

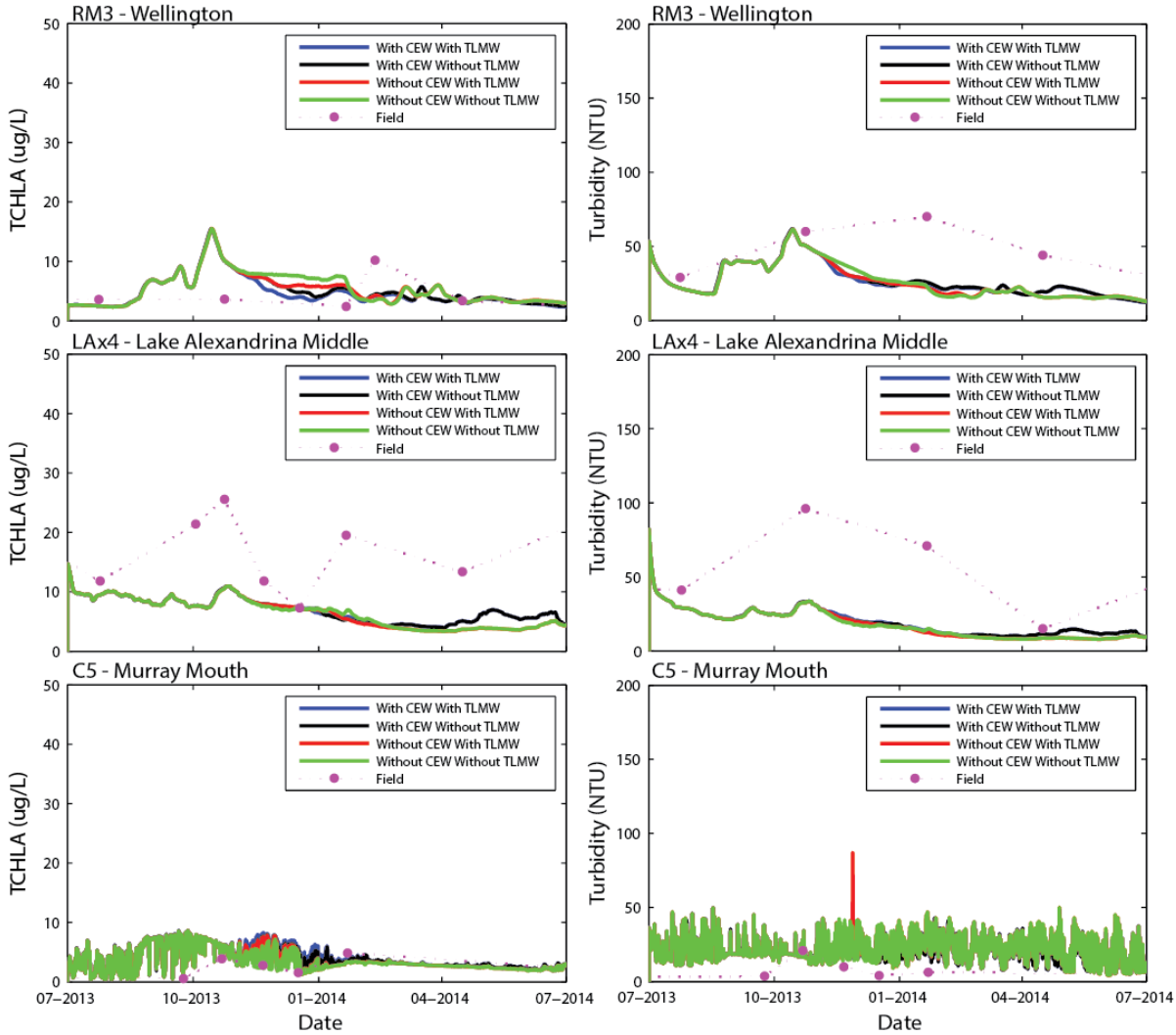


Figure 35. Observed and modelled (with and without environmental watering) chlorophyll *a* concentrations and turbidity with and without environmental flows. Scenarios include with and without both commonwealth environmental water (CEW) and The Living Murray water (TLMW).

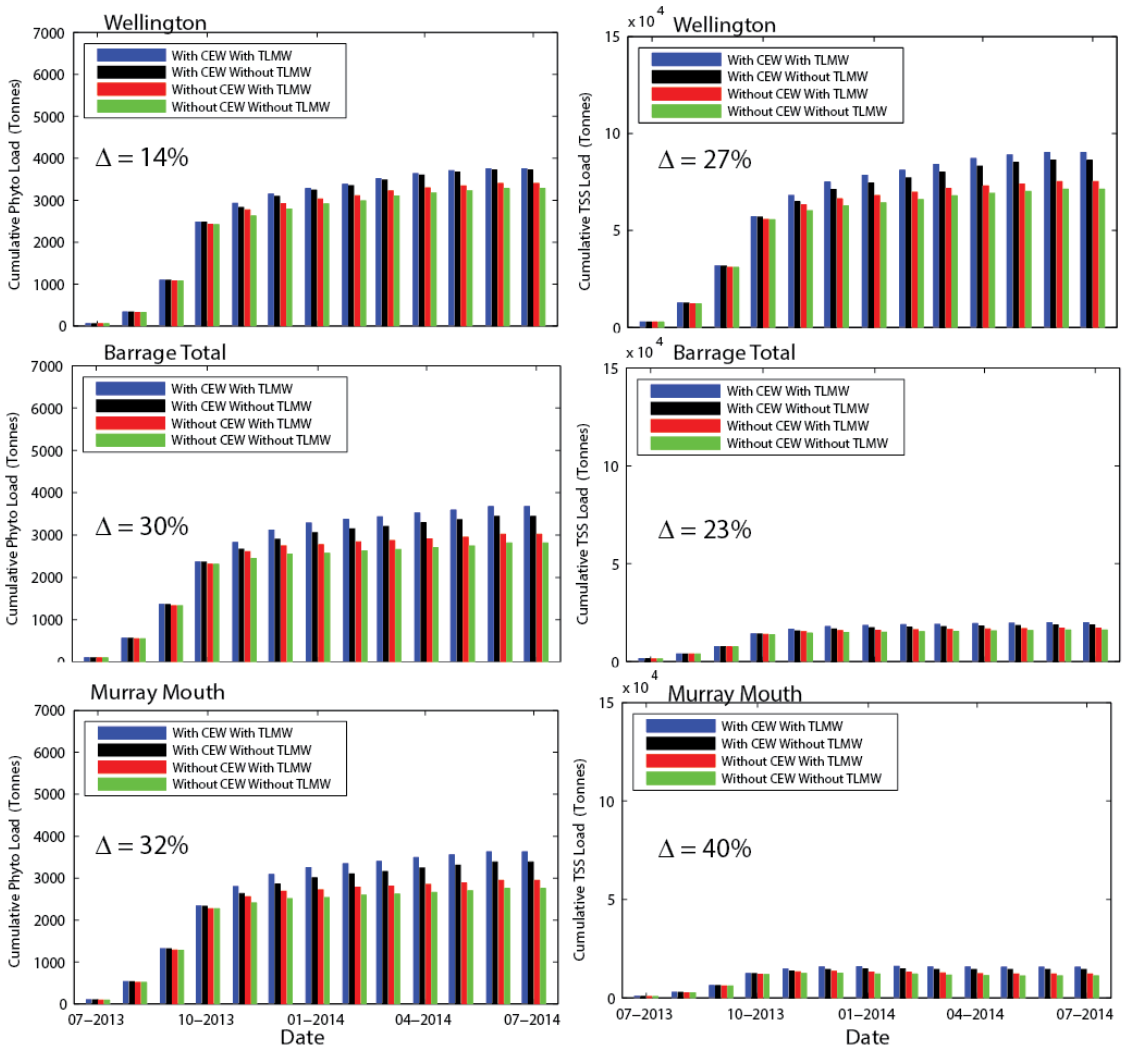


Figure 36. Modelled cumulative phytoplankton (as measured by carbon) and total suspended solid (TSS) exports (net) with and without environmental water delivery. Scenarios include with and without both commonwealth environmental water (CEW) and The Living Murray water (TLMW). Δ is the percentage increase in loads associated with environmental water provisions (CEW and TLMW).

4.4 Coorong (modelling)

Coorong hydrodynamic modelling

Model results for salinity and water level for the whole period 1963–2014 are shown in Figure 37. Since we mainly examined thresholds for habitat modelling, the results presented for a water level threshold of +0.2 m AHD and a salinity threshold of 85 g L^{-1} are exemplified. While water level will modify outcomes of habitat modelling the salinity tolerance of species is the main driver in habitat availability for individual species. The threshold value for the scenario presented in Figure 37 is indicative of juveniles of fish species with higher salinity tolerances, which could provide a preliminary understanding of habitat suitability.

Changes in water level and salinities during 2013-14 due to environmental watering are shown in Figure 38. Compared to the reference scenario (with environmental watering from all sources), differences for Scenario 2 (with Commonwealth environmental watering only) were low but could become large for Scenario 3 if there was no environmental watering. Larger changes in both water levels and salinities were due to the greater contribution of Commonwealth environmental water in the summer months. Overall, the impact of missing environmental watering on salinities seems to be greatest in the North Lagoon due to lack of a freshening effect from barrage inflows whereas the effect on water level seems to be more significant in the South Lagoon due to the decreased water inflow from the North Lagoon.

As in 2012-13, environmental watering had a strong impact on water level and salinity in 2013-14. Without environmental water a loss of up to 150 GL $month^{-1}$ and 175 GL $month^{-1}$ (see inset in Figure 38) for the scenarios with CEW and without TLM, and without either, respectively, would have the capacity to reduce water levels by about 6 cm to 9 cm and increase salinity by about 10 g L^{-1} to 15 g L^{-1} in the North Lagoon. The effect on salinity would be larger in 2013-14 than in 2012-13 since the missing flow conditions would occur over a comparatively larger time period.

a)

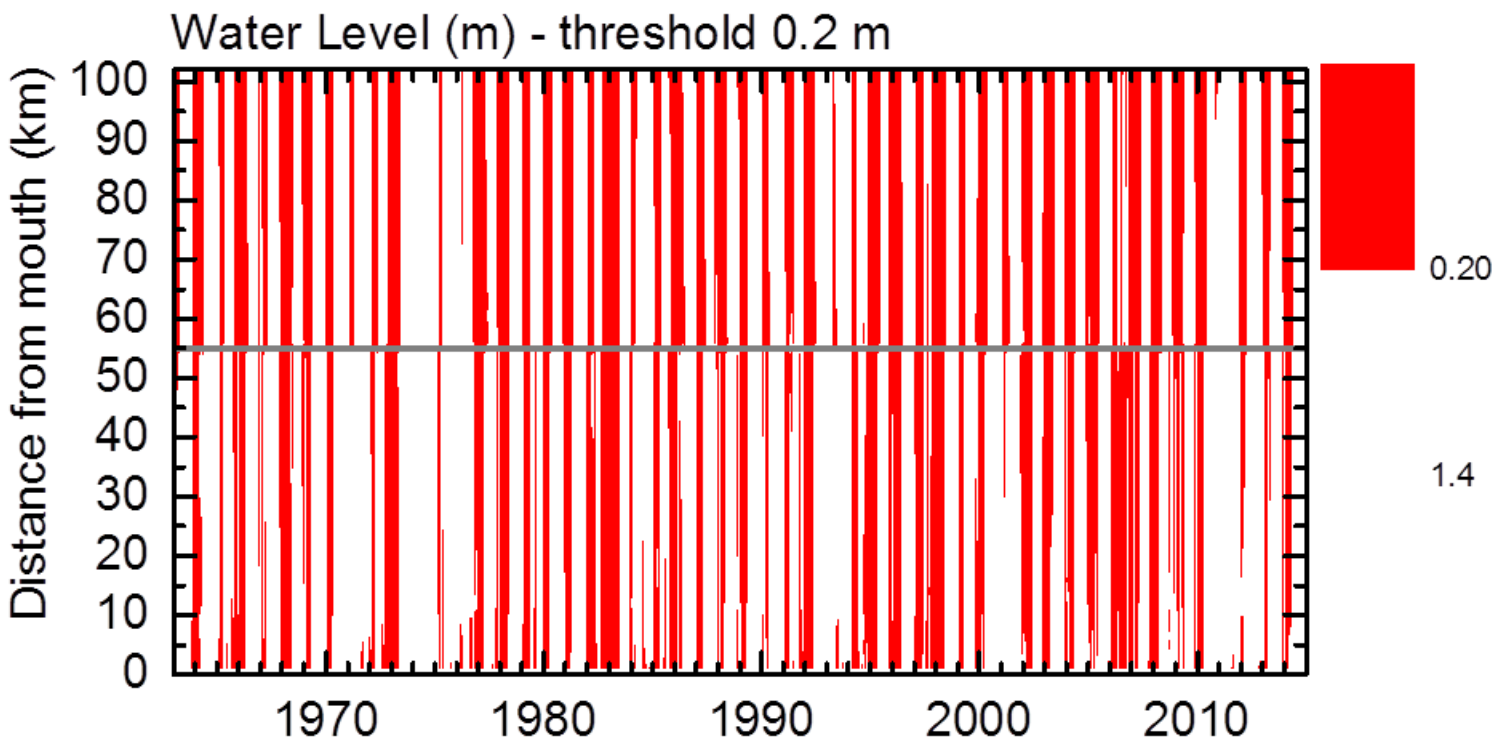


Figure 37. Water level (m AHD) and salinity (g L^{-1}) in the Coorong as modelled with CHM v2.1.0 for the full simulation period from 1963 to 2014 along the Coorong from mouth (0 km) to Salt Creek (102 km). Black horizontal line at 55 km delineates North and South Lagoon. a) Water level visualised for a threshold of 0.2 m AHD, red areas indicate time periods when water level was ≤0.2 m AHD;

b)

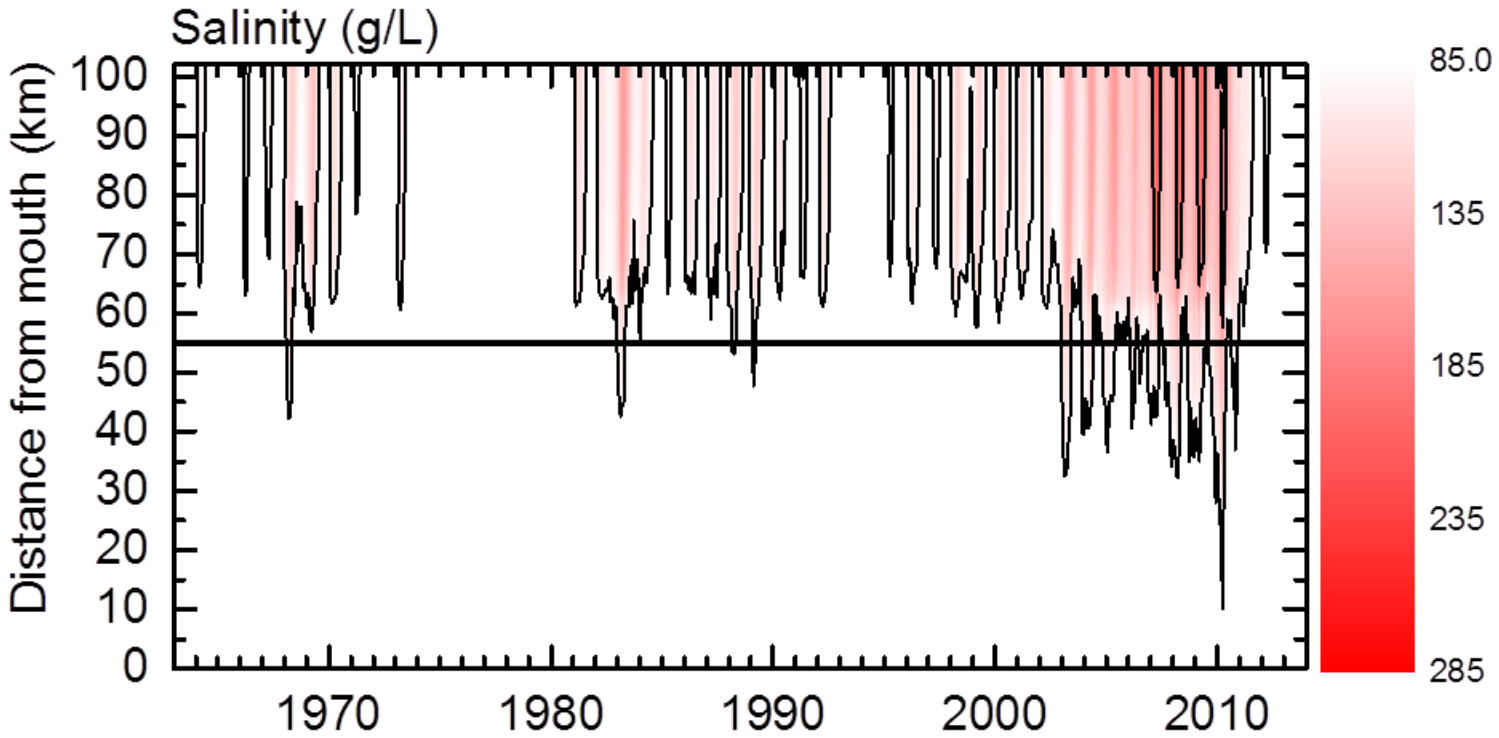


Figure 37 continued. Water level (m AHD) and salinity (g L^{-1}) in the Coorong as modelled with CHM v2.1.0 for the full simulation period from 1963 to 2013 along the Coorong from mouth (0 km) to Salt Creek (102 km). Black horizontal line at 55 km delineates North and South Lagoon. b) Salinities above 85 g L^{-1} (black contour line) to be compared with fish habitat simulations.

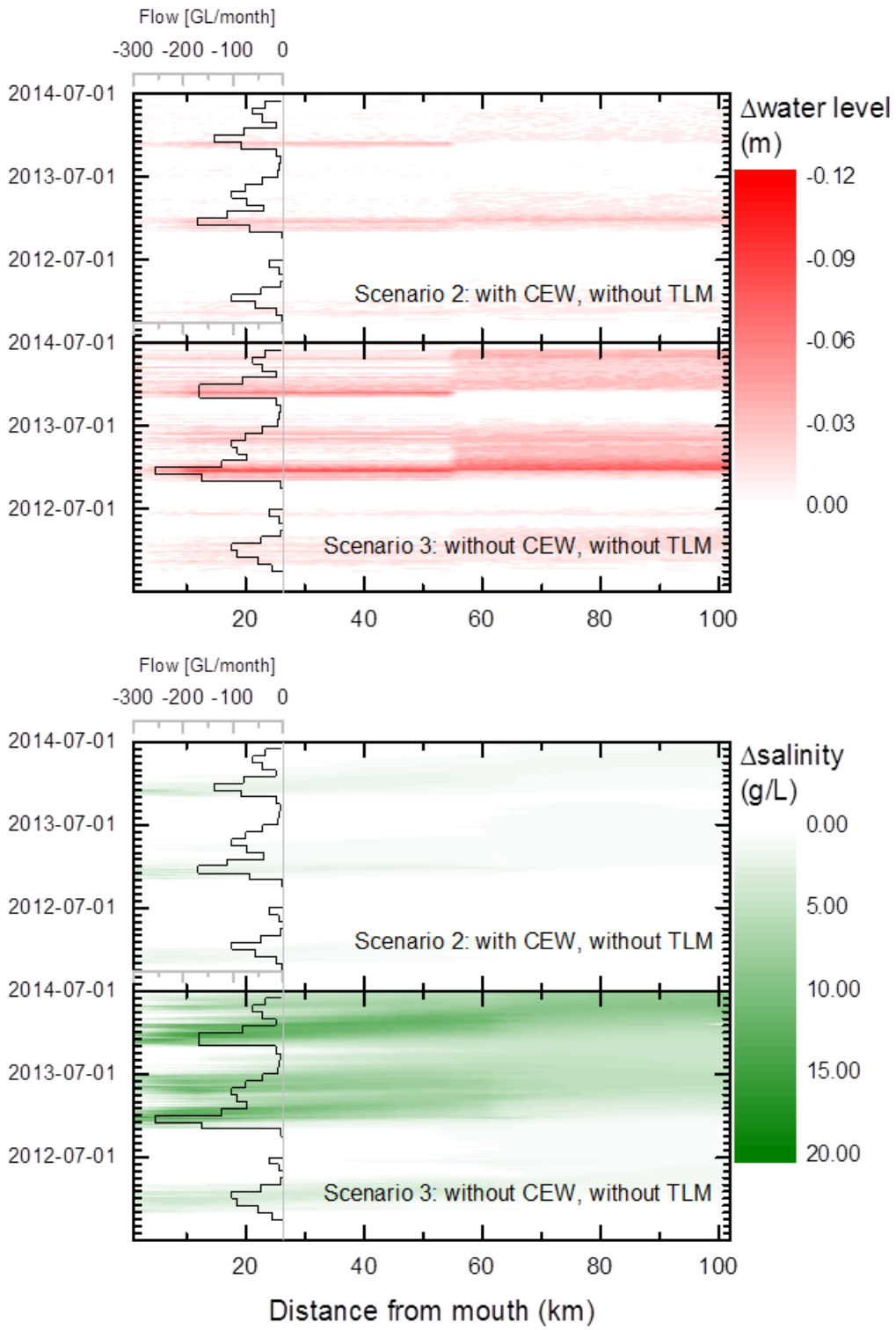


Figure 38. Changes in water level (upper graph) and salinity (lower graph) for scenarios 2 (with CEW, without TLM) and 3 (without CEW, without TLM) with respect to the reference (with CEW and TLM) for the period July 2012 to June 2014.

Ruppia tuberosa

Ruppia tuberosa simulation 1963 to 2013

The modelled probability of *R. tuberosa* sediment propagule bank replenishment between 1963 and 2013 in the South Lagoon of the Coorong between Parnka Point (55 km from the Murray Mouth) and Salt Creek is shown in Figure 39. Over the 51 year model simulation period, there was a general decline in the probability of replenishment (Figure 39).

During the early to mid-1960s, the probability of replenishment was mostly high in the South Lagoon but this was followed by a period of low probability of replenishment in the late 1960s (Figure 39). During the 1970s, the probability of replenishment was high, except for two periods of approximately one year in 1972 and 1977 (Figure 39). During the 1980s, the duration of the periods of hydrological conditions that resulted in low probability of replenishment increased and periods of high probability of replenishment decreased (Figure 39). A further increase in the duration of unfavourable hydrological conditions was observed in the 1990s, with only three out of 10 years having hydrological conditions that would have resulted in a probability greater than 25% (Figure 39). From 2001 to 2011, the probability of replenishment was lower than 25% for the South Lagoon, except in 2001 in the northern 30 km of the lagoon (Figure 39). The 2010-11 flood resulted in favourable hydrological conditions for *R. tuberosa* in the South Lagoon with consequently higher probability of replenishment (Figure 39). Favourable conditions for *R. tuberosa* have since prevailed and 2013 showed a full replenishment probability in the whole lagoon (Figure 39).

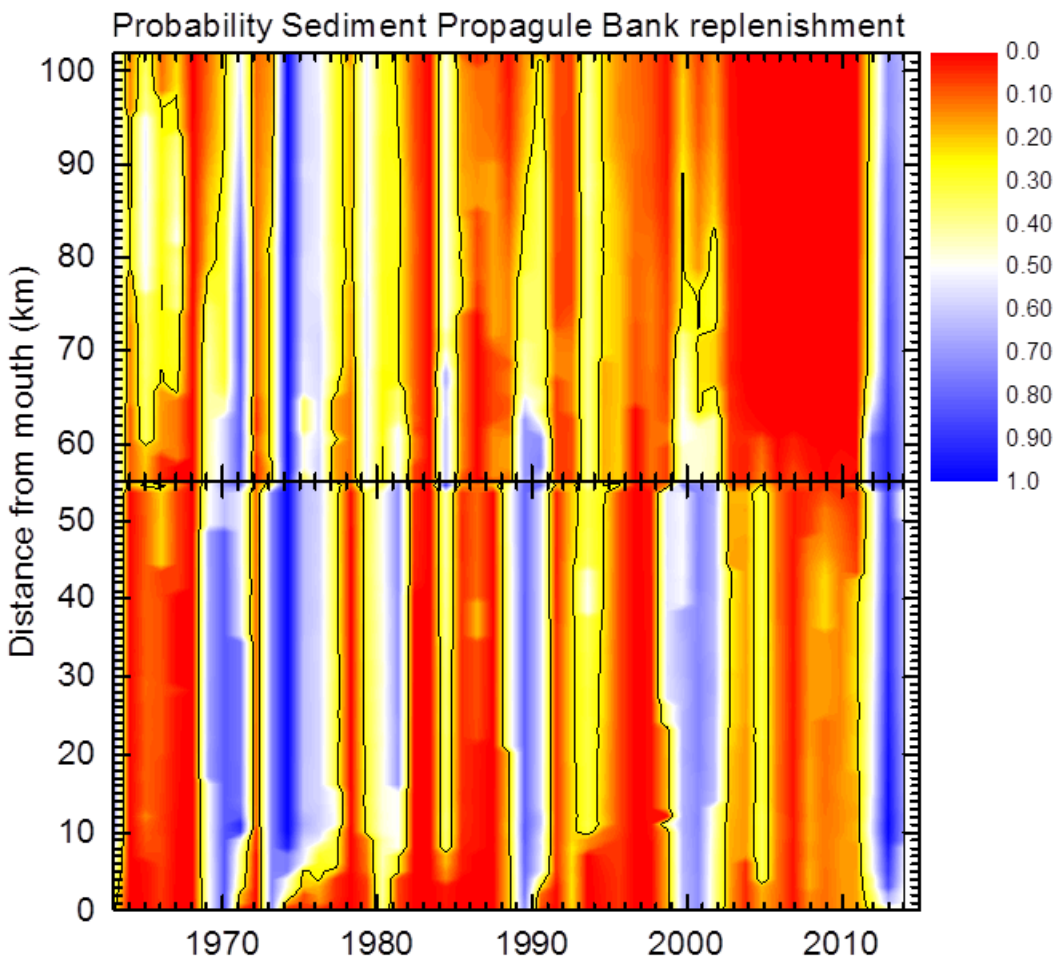


Figure 39. Model output showing the probability of *Ruppia tuberosa* sediment propagule bank replenishment from 1963 to the end of 2013 in the North Lagoon (lower graph) and South Lagoon (upper graph) of the Coorong between Murray Mouth channel (0 km) and Salt Creek (102 km from the Murray Mouth). The contour line on the plot represents the 25% probability of sediment propagule bank replenishment.

Ruppia tuberosa environmental watering effects for 2011–2013

In addition to the simulations of *R. tuberosa* sediment propagule bank replenishment over a 51 year period, the changes due to environmental watering and thus water level and salinity conditions were simulated for 2011 until 2013 for full calendar years. The reference condition refers to actual barrage outflows (natural flow plus Commonwealth environmental water and TLM environmental water), scenario 2 refers to natural flows plus Commonwealth environmental water only and scenario 3 is modelled flow without environmental water. Since the *R. tuberosa* response model is defined on calendar years, sediment propagule bank replenishment probabilities

could only be simulated for the full calendar years 2012 and 2013, while the simulations for 2011 include only changes in barrage flow scenarios from July 2011 onwards as no data for environmental water splits were available prior to this date. Thus the simulations for 2011 are biased towards relatively low effects since there was no year round information on environmental watering available. The sediment propagule bank replenishment probabilities together with changes due to environmental watering are shown in Figure 40. In 2011, there was an increased probability of propagule bank replenishment throughout the Lagoon compared to the previous drought years (Figure 40a). As no information on environmental watering was available for the start of this year, there is no significant difference between different watering scenarios. The propagule bank replenishment probability ranged between 25% and 50%. In 2012, the probability of propagule bank replenishment has significantly increased to values above 75% (Figure 40b). The addition of Commonwealth environmental water increased the chance of propagule bank replenishment up to 4% in the northern 30 km of the South Lagoon (Figure 40b).

In 2013, propagule bank replenishment was simulated between 50% and 75% (Figure 40c). A major effect of the high replenishment rates in the North Lagoon can be attributed to the addition of Commonwealth environmental water, which increased the chance of propagule bank replenishment up to 20%. However, the addition of environmental water made very little difference to the chance of propagule bank replenishment in the South Lagoon (Figure 40c). The noticeable jumps in the curves are a result of probabilities related to thresholds in this analysis, i.e. actual values of salinity or water level were sometimes just below or sometimes just above a given threshold resulting in low or high replenishment probability.

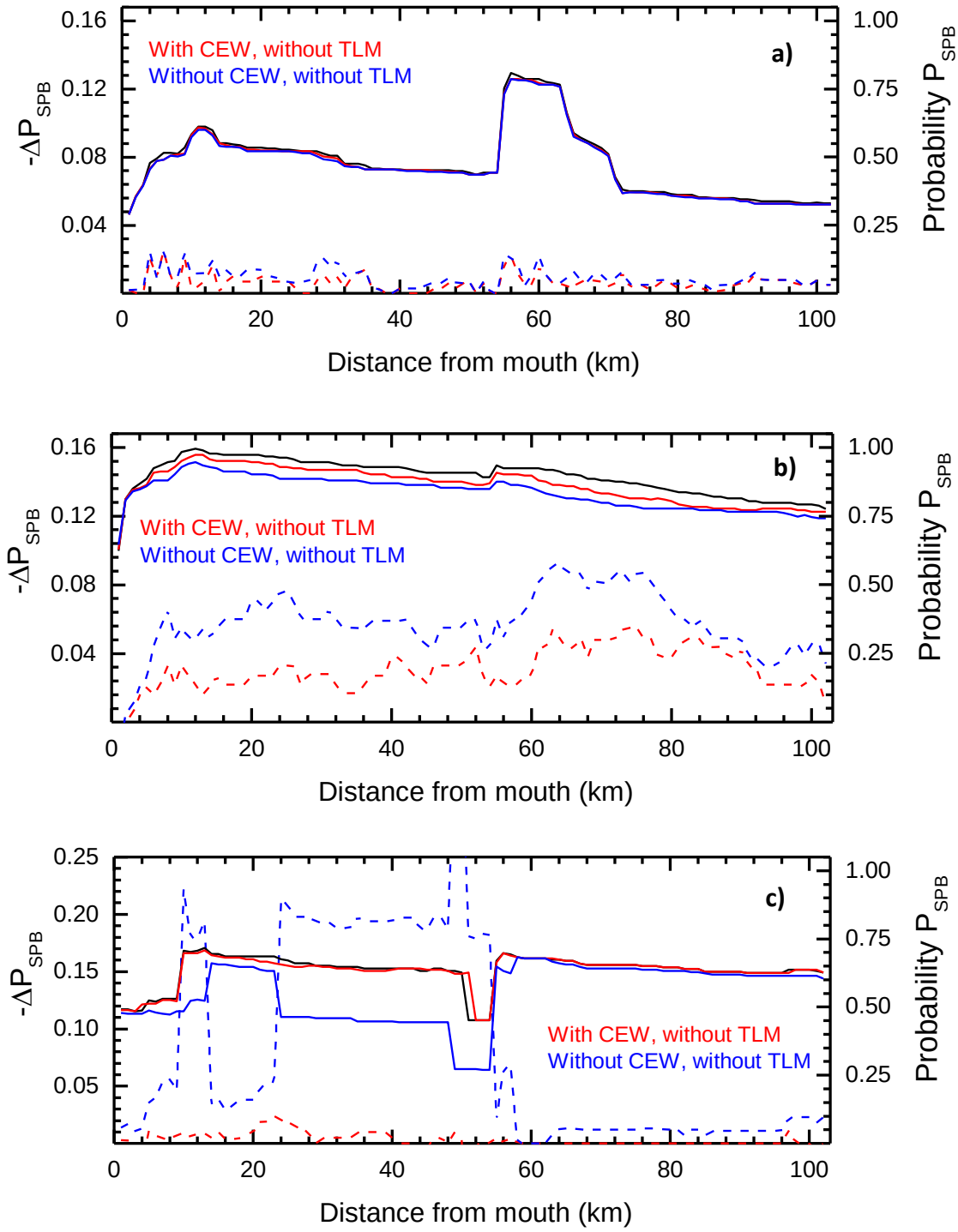


Figure 40. Sediment propagule bank replenishment probability for *Ruppia tuberosa* for a) 2011, b) 2012 and c) 2013. The black line represents the reference simulation including both, CEW and TLM water. Simulations with CEW and without TLM (red) and without CEW and TLM (blue) are given in terms of probability values for sediment propagule replenishment (right axis, solid lines) as well as changes with respect to the reference (dashed lines). The year 2011 does not show large differences as changes in scenarios were only effective from July 2011.

Fish habitat

Fish habitat simulation 1963 to 2014

In total, seven fish species were included in the simulations using the salinity tolerance thresholds for warm and cool periods (Table 4). The results of the annual probability analysis in the reference scenario from 1963 to 2014 are shown in Figure 41. The results for the simplified threshold analysis generally agree with the annual exceedance probability analysis shown in Figure 41. Both daily and annual analyses show that during the mid-1970s the entire Coorong provided suitable fish habitat in terms of salinities for all seven species, whereas from early 2000 to 2010 almost all these species were excluded from the South Lagoon. For the species with lowest salt tolerance (i.e. mulloway and Tamar goby), habitat suitability (percentage of available habitat) was less than 20% in the South Lagoon for most of the years from 1963 to 2013. In contrast, for the more salt tolerant smallmouthed hardyhead, habitat was suitable in the South Lagoon throughout the years, except during the millennium drought. The update to June 2014 shows that since 2012 the habitat suitability for all fish species has substantially increased; whereas the suitability for mulloway, the least salt tolerant species, remained low in the South lagoon.

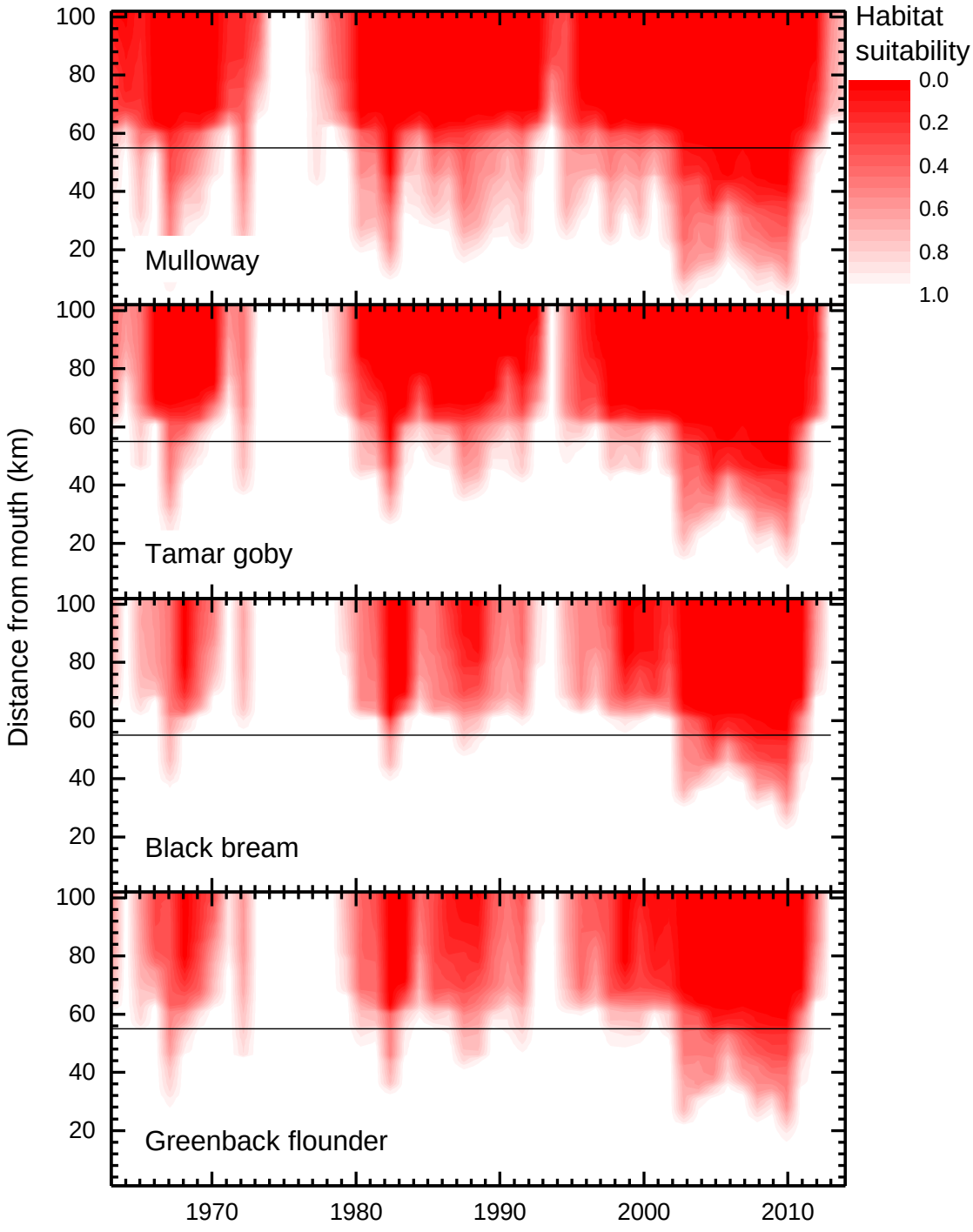


Figure 41. Habitat suitability for fish species along the Coorong from mouth (0 km) to Salt Creek (102 km) calculated for the whole simulation period 1963-2013 on an annual basis using salinity tolerance thresholds. The horizontal black line indicates the border between North (0-55 km) and South Lagoon (55-102 km). Note that as the fish simulation starts mid-year, a plotted value for the year 2013 would correspond to the period July 2013 until June 2014, etc.

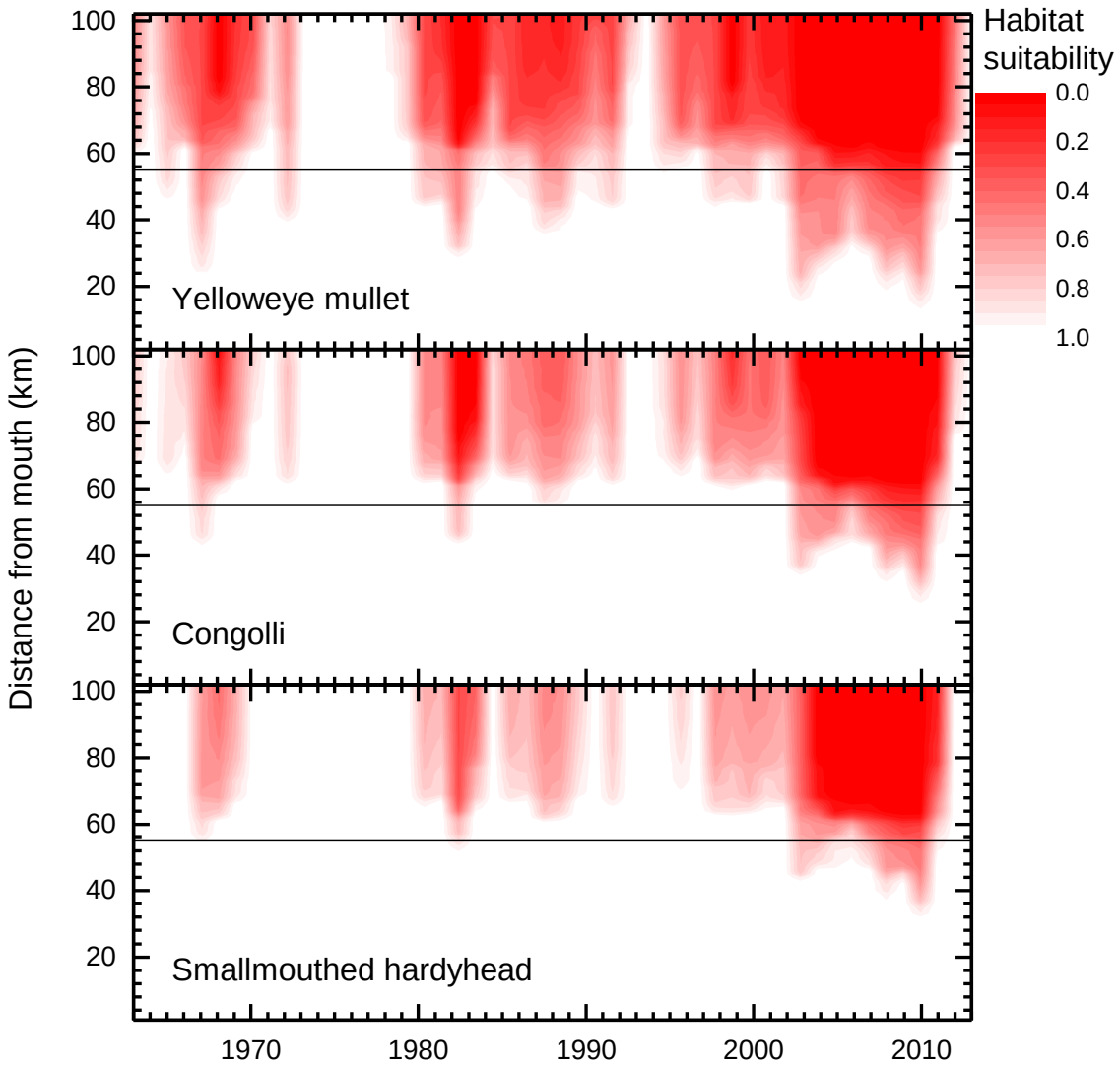


Figure 41 continued: Habitat suitability for fish species along the Coorong from mouth (0 km) to Salt Creek (102 km) calculated for the whole simulation period 1963-2013 on an annual basis using salinity tolerance thresholds. The horizontal black line indicates the border between North (0-55 km) and South (55-102 km) Lagoon. Note that as the fish simulation starts mid-year, a plotted value for the year 2013 would correspond to the period July 2013 until June 2014, etc.

Fish habitat environmental watering effects for 2013-14 (July to June)

Different plots for changes due to environmental watering according to Scenarios 2 (with CEW, without TLM) and 3 (without CEW and TLM) are given in Figure 42 for the annual habitat suitability. However, only three out of the seven species, mulloway, Tamar goby and yelloweye mullet, showed changes in suitable habitat due to altered salinity in the South Lagoon for 2013-14. The salinity thresholds of the other species were always below the simulated salinity levels along the Coorong, i.e. always classified as suitable habitat. Taking mulloway as the fish species most susceptible to changes in salinity, there would have been a reduction up to 3% (13%) in habitat suitability if less (or no) environmental water was delivered during 2012-13, i.e. for Scenarios 2 (with Commonwealth environmental watering only) and 3 (with no environmental watering), respectively (see Ye *et al.* 2015b). Environmental conditions in 2013-14 were more favourable to these salt tolerant fish species because habitat suitability increased from about 50% in 2012-13 to 75% in 2013-14 for the southernmost part of the South Lagoon. The addition of environmental water accounts for an increase of 25% in habitat suitability, a comparable value as for *Ruppia tuberosa* habitat probability.

Threshold analysis was also used to investigate the extent, or the retreat, of fish habitat under barrage flow scenarios, i.e. differences between results for Scenarios 2 and 3 and the reference scenario. The retreat/extent of the three impacted fish species is shown in Figure 43. For mulloway, Scenarios 2 and 3 yielded a retreat in suitable fish habitat of 7 km and 30 km, respectively, during the cooler months from April to June 2013 (Ye *et al.* 2015b). The simulation for 2013-14 shows, that the addition of environmental water (Scenario 3: without CEW, without TLM) increased the habitat reach for mulloway by up to 40 km along the Coorong lagoons from January to May.

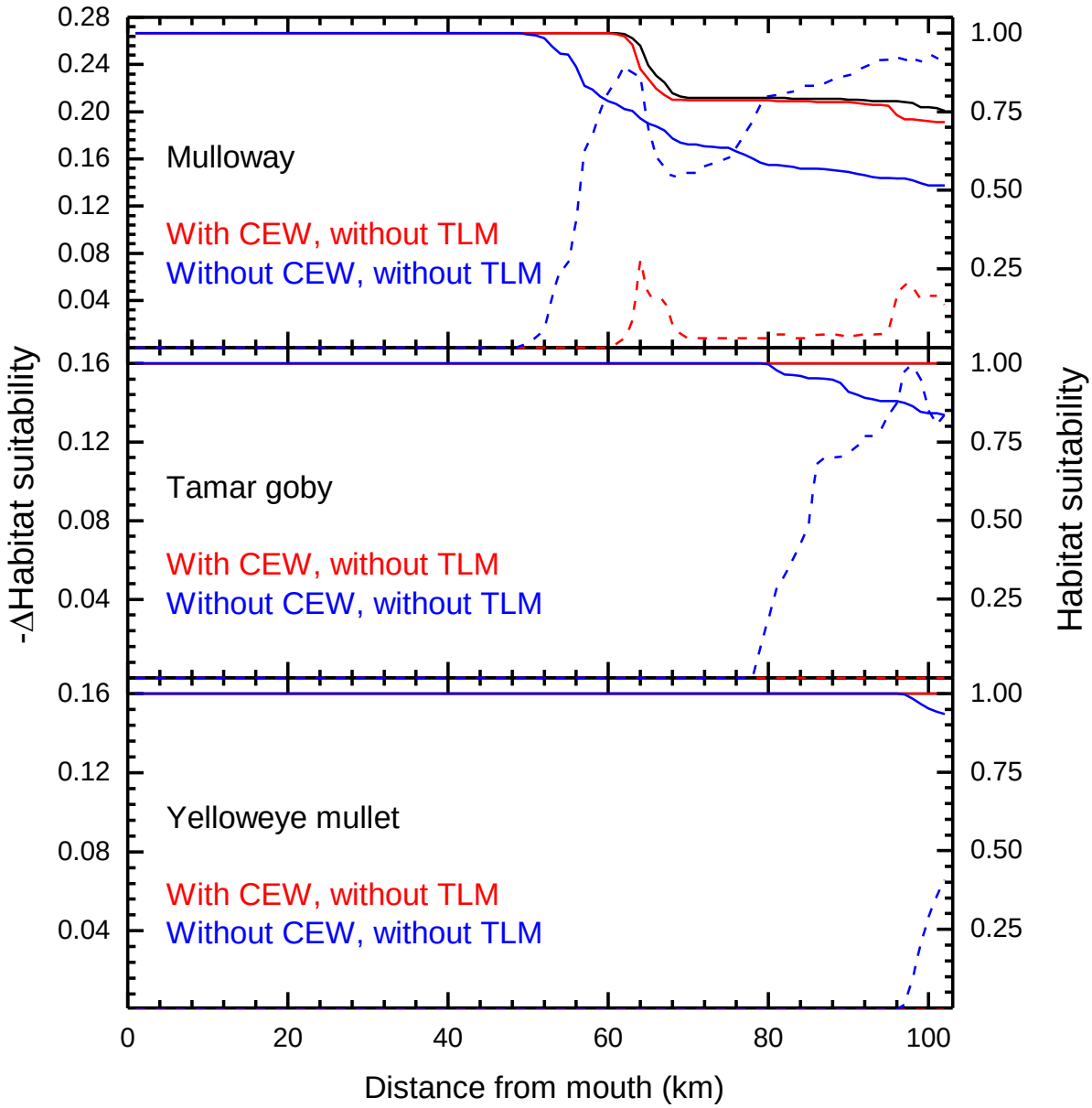


Figure 42. Changes in fish habitat suitability assuming less barrage flow in 2013-14 with respect to the reference simulation (including CEW and TLM water). The black line represents the reference simulation including both, CEW and TLM water. Simulations with CEW and without TLM (red) and without CEW and TLM (blue) are given in terms of probability values for fish habitat suitability (right axis, solid lines) as well as changes with respect to the reference (dashed lines). Species not shown did not experience changes in this particular year. Sign of changes in habitat suitability chosen in a way that a positive value in the habitat change signifies a loss.

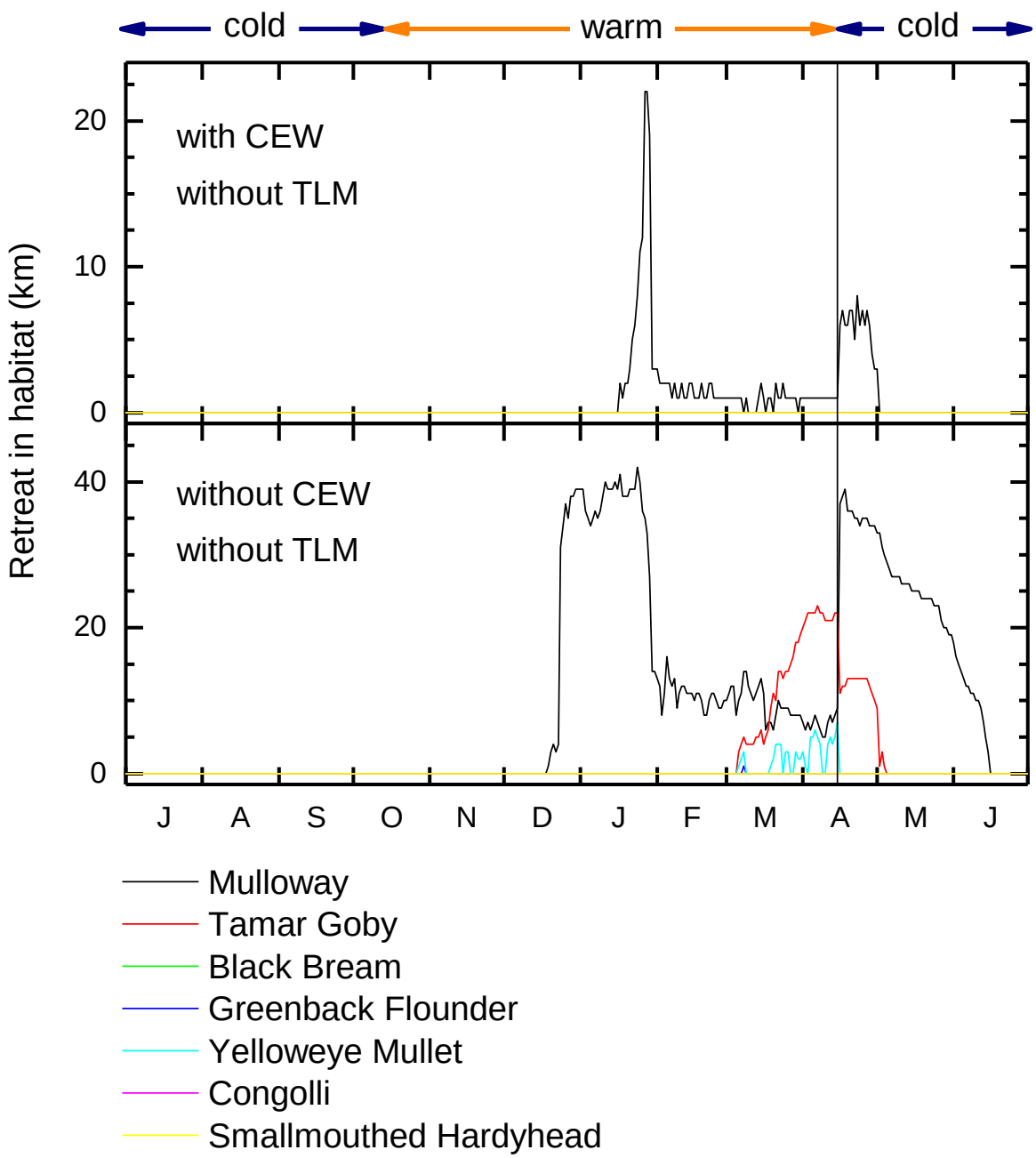


Figure 43. Differences in habitat reach with respect to the reference simulation (including CEW and TLM water) for 2013-14. Upper graph shows the scenario result with CEW and without TLM water, while the lower graph shows the scenario results including neither CEW nor TLM water. Vertical black lines delineate cool from warm periods where different thresholds were used. Only three of the seven species (mulloway, Tamar goby and yelloweye mullet) for which simulations were performed showed changes in this specific year.

5 DISCUSSION AND EVALUATION

5.1 Golden perch spawning and recruitment

In 2014, golden perch populations in the floodplain and gorge geomorphic regions of the lower Murray River were dominated by age 4+ fish, representing 64% and 45% of the sampled populations, respectively. In the floodplain geomorphic region, the remainder of the population comprised predominantly young fish (i.e. age 2+, 6%, and 3+, 14%) with low proportions (≤4%) of age 0+, 8+, 13+ and 18+ fish. In the gorge geomorphic region, the remainder of the population mostly comprised older fish (i.e. age 13+ and 17+), with smaller numbers of age 3+ fish.

Overall, our data demonstrate episodic recruitment of golden perch during the period of the Millennium drought (2001–2010) but more consistent recruitment from 2009 onwards. Consecutive year-classes (i.e. age 0+–4+) from 2010–2014 were spawned in association with within-channel and overbank increases in flow in the lower Murray River and the Darling River. The addition of these year classes has improved the resilience and hence health of golden perch populations in the lower Murray River.

In 2014, age 0+ (YOY) fish, spawned in 2013-14, were collected in low abundances from the floodplain and gorge geomorphic regions of the lower Murray River where they comprised 4% and 2% of the sampled populations, respectively. Larval golden perch and YOY recruits collected in the lower Murray River in 2013-14 were spawned over a broad period from late October to late December 2013. Otolith core Sr isotope ratios indicate that the majority of these fish (94%, n = 17) were spawned in the lower Murray River, downstream of the confluence with the Darling River. One fish, however, spawned in early December 2013, had an otolith core Sr isotope ratio which was consistent with having originated from the Darling River.

Golden perch spawned in the lower Murray River were generally spawned in association with flow recession following a within-channel flow peak of ~25,000 ML day^{-1} and subsequently elevated within-channel flows, combined with water temperatures >19 °C. Fish were spawned prior to (i.e. mid to late October 2013), and during, the delivery of environmental water (i.e. November to late October 2013). The

one YOY golden perch originating from the Darling River was spawned in association with a ~2,500 ML day^{-1} within-channel increase in flow in the Darling River in early to mid-December 2013. These findings support the results of several studies (e.g. Zampatti and Leigh 2013; Zampatti *et al*. 2015) that water management, or unregulated flows, that promote flow variability (within-channel and overbank) above regulated entitlement flows, may stimulate golden perch spawning in the lower Murray and Darling Rivers subsequently promoting golden perch recruitment (to at least YOY) in the lower Murray River. Furthermore, sequential years of flows greater than regulated entitlement flow may result in successive year classes of fish, thus improving the resilience (i.e. health) of golden perch populations in the lower Murray River.

The results of this study support the hypothesis that flows >15,000 ML day^{-1} in the lower Murray River during summer would result in recruitment (to age 0+) of golden perch in the South Australian reaches of the Murray River. In 2013-14, otolith microstructure and chemistry indicate that the majority (94%) of larval fish and YOY recruits were spawned in the lower Murray River downstream of the Darling River, whilst one YOY recruit was spawned in the Darling River. Furthermore, many (83%, n = 15) of these fish with known spawn date were spawned in the lower Murray River over the period when environmental water was delivered. Thus the delivery of Commonwealth environmental water in 2013-14 likely contributed to the CEWO objective of providing a flow regime that supports breeding of native fish species and recruitment of juvenile life stages (CEWO 2013).

5.2 Lateral movements of fish

Connectivity between wetlands and main channel habitats has long been recognised as playing a significant role in the health and sustainability of fish assemblages (Junk *et al*. 1989). The dynamics that drive relationships between environmental factors and fish populations are complex and have been the subject of recent studies in Australia (e.g. Conallin *et al.* 2011; 2012; Lyon *et al.* 2010) and globally (e.g. Lasne *et al.* 2007; Gorski *et al.* 2014). Of particular interest in highly modified systems, like the Lower Murray River, is the relationship of flow driven variables that may allow for improvement of fish assemblages towards increased native species through effective management of hydrological conditions.

The Lower Murray River has been described as a "homogenous series of lentic environments under low flows" (Zampatti and Leigh 2013a), due to fragmentation of habitat through a series of low level weirs. These contiguous weir pools, however, are usually managed independently and therefore can have contrasting environmental characteristics which are not always acknowledged. During this study which occurred in wetlands associated with two adjacent weir pools, we found that the relationship between main channel flow and weir pool level wasn't always positively related under the flow regime in 2013-14. Furthermore, no positive relationship was found between inlet height (water levels of inlet channels that connect the wetlands with the main channel) and main channel flow for both wetlands. A lack of consistency in relationships among such hydrological factors suggests in turn that relationships between environmental variables and fish assemblages may be highly dependent on particular circumstances of the wetland of interest. We propose that using flow in the main channel as a surrogate for hydraulic and environmental features in off-channel habitats may result in confounding effects. Furthermore, assumptions of biotic responses within wetlands to changes in main channel flow should be tested before subsequent use in environmental models. Lack of testing may lead to flawed predictions of biotic responses and the ensuing risk of not obtaining the desired ecological outcomes through environmental watering.

Generally, fish assemblages moving in and out of wetlands was driven by inlet flow, flow in the main channel and water temperature. It must be noted that these variables were correlated with several other variables excluded from the statistical analyses, but that may have a biological significance that cannot be identified through this type of study. The three environmental variables (i.e. inlet flow, temperature and flow in the main channel) strongly influenced fish movement, but the patterns of response differed among species in different wetlands. Variability was partially due to very distinct fish assemblages between wetlands with Gambusia, a small-bodied exotic species, numerically dominating Martin's Bend, whilst carp gudgeon, a small-bodied native species, was numerically abundant in Hart Lagoon.

Patterns of lateral fish movement were complex and highly variable among flow phases and changes were not consistent among wetlands. Spatial variability is a common attribute of fish assemblages in freshwater systems (Jackson *et al.* 2001) and the observed variability in this study was not unexpected given the substantial

variability in the physical, biological and hydrological attributes of wetlands in the Lower Murray River. Lateral movement of fish was highly variable between wetlands even at the individual species level. There was continuous bi-directional movement of fish throughout the study, with more movement detected for both wetlands in the last sampling event on the decreasing limb of the environmental water delivery. During this time, flow in the main channel was the lowest of the study period.

In 2013-14, changes in flow in the main channel, due to the delivery of environmental water, were short-lived and of a relatively small magnitude. Furthermore, the response of water quality and other environmental variables to environmental water delivery were highly variable and inconsistent. The expected positive relationship between flow and water level in the main channel and of those with inlet height and inlet flow was not observed. This may be due to the fact that the magnitude of the environmental flow during 2013-14 was not sufficient to have noticeable effects on environmental variables measured. The differential management of weir pools within the Lower Murray River makes it difficult to assess the effects of environmental watering on fish lateral movement as a whole and caution must be exercised when assuming that small changes in main channel flow will have detectable effects on off-channel habitats.

Hart Lagoon

Movement within Hart Lagoon inlet was lower during the stable flow phase than during both phases of environmental watering. Increased movement during environmental flow phases was bi-directional and for several species, suggested that variability in hydraulic conditions encouraged fish movement, which is consistent with observations from the upper Murray River (Lyon *et al*. 2010). Carp gudgeon was the numerically dominant species, which was expected as many small-bodied species are known to use the off-channel habitats as they offer increased habitat diversity, survival, feeding and reproduction opportunities (Balcombe and Humphries 2006). The second most abundant species using Hart Lagoon inlet was bony herring, which is a medium-sized, native species with an opportunistic life-history. Consistent with findings in Conallin *et al*. (2012), bony herring showed increased movement with increasing water temperature (i.e. SE3 and SE4). Exotic species were not prevalent in Hart Lagoon until the final sampling event during the decreasing environmental water

phase when a cohort of small carp was collected entering the wetland. This species is known to actively seek off-channel habitats as refuges and nursery grounds (Stuart and Jones 2006).

Martin's Bend

Movement in Martin's Bend was characterised by more fish moving into than out of the wetlands. The main contributors to this trend were the exotic Gambusia, and the native bony herring and carp gudgeon. Very little variation in CPUE was detected in the different flow phases, however, fish assemblages in the descending environmental water phase was distinct from the other. During this last phase, trends markedly changed with exotic Gambusia noticeably exiting and carp entering the wetland.

Martin's Bend was utilised consistently by a broad range of native species including some large-bodied species such as golden perch, silver perch and freshwater catfish. Native fish species tended to move more into than out of the wetland, however, presence and abundance within the wetland were low or nil suggesting that the inlets itself may provide a type of habitat with favourable conditions. Maintenance of connection channels may be important not only because of the hydrological functionality in linking off-channel and within-channel environments but also due to the potential biological interactions that occur in this habitat.

5.3 Dissolved and particulate material transport

The approach used for this study successfully evaluated changes in concentrations and transport of dissolved and particulate matter associated with environmental water delivery. Some differences in observed and modelled data were evident but further refinement of the model will continue to reduce uncertainty in model outputs, particularly for the more sensitive parameters (dissolved nutrients and chlorophyll). The modelling outputs suggest that during 2013-14 environmental watering resulted in significant increases in the transport and export of salt, nutrients, chlorophyll (phytoplankton) and suspended solids through the Lower Murray River, Lower Lakes and Murray Mouth. The transport of this dissolved and particulate matter has an important role in provision of habitat for biota and resources that maintain the productivity of downstream ecosystems (Cook *et al*. 2010). Thus, environmental flows in the Lower Murray can influence processes that are essential for providing habitat

and resources for aquatic biota, further supporting the findings of Aldridge *et al*. (2013), Ye *et al*. (2014) and Brookes *et al.* (2015).

The majority of the total exports were achieved during an unregulated flow event prior to a period of environmental water delivery (July to October). Thereafter, however, the additional exports were almost entirely associated with environmental provisions and overall environmental water contributed up to 45% of total exports of dissolved and particulate matter. It was evident that both sources of environmental water made contributions to the transport and export of dissolved and particulate matter, although a majority tended to be associated with commonwealth environmental water due to the greater volumes of water provided. On occasions, the combined contribution of both sources was greater than that of the individual sources alone.

It was evident that the increase in transport of dissolved and particulate matter associated with environmental watering was largely a result of the increased discharge rather than altered concentrations. An exception to this was salinity within the Murray Mouth and to a lesser extent the Lower Lakes, with lower salinities associated with environmental water delivery. This is consistent with previous studies (see Aldridge *et al*. 2012; Mosley *et al*. 2012; Aldridge *et al*. 2013) that have suggested environmental water deliveries to the Lower Lakes and Coorong during periods of low to moderate flows are important for reducing salinity levels. Reductions in salinity levels provide an important functional role by providing habitat for aquatic organisms (e.g. fish habitat in the Coorong).

5.4 Coorong (modelling)

Ruppia tuberosa

For periods where there is information regarding the distribution and abundance of *R. tuberosa* in the South Lagoon (from the mid-1970s onwards) there is generally a good correlation between the model output and what was reported in the literature. During the mid to late 1970s when *R. tuberosa* was abundant and widespread in the South Lagoon (Womersley 1975; Geddes and Brock 1977; Gilbertson and Foale 1977; Brock 1979; Brock 1981) the probability of sediment propagule bank replenishment was high, except for two periods of approximately one year in 1972 and 1977 (Figure 39). These relatively short periods when there was a low probability of replenishment

probably had little impact on the population dynamics of *R. tuberosa* because this species has a persistent seed bank (i.e. not all of the seed in the sediment seed bank germinates at one time) (*sensu* Thompson and Grime 1979) and the unfavourable periods were followed by extended periods of favourable conditions when there was a high probability of propagule bank replenishment (Figure 39).

During the 1980s, *R. tuberosa* was still abundant and widespread in the South Lagoon of the Coorong (Paton 1982; Geddes and Butler 1984; Geddes 1987); however, the duration of the periods of hydrological conditions that resulted in low probability of propagule bank replenishment increased and periods of high probability decreased (Figure 39). The periods of high probability of propagule bank replenishment, whilst relatively short, appeared to be sufficient to maintain the *R. tuberosa* population.

A further increase in the duration of unfavourable hydrological conditions was observed in the 1990s, with only three out of 10 years having hydrological conditions that would have resulted in a probability greater than 25% of propagule bank replenishment (Figure 39). *R tuberosa* was widespread and abundant during the early 1990s (Leary 1993; Paton 1996; Nicol 2005), which corresponds to a period of high probability of propagule bank replenishment (Figure 39). However, by the late 1990s, the distribution and abundance of *R. tuberosa* was showing signs of decline particularly at the southern end of the South Lagoon (Freebairn 1998; Paton 2000; Nicol 2005; Whipp 2010), which corresponded with a period of low probability of propagule bank replenishment (Figure 39).

From 2001 to 2011, the probability of *R. tuberosa* sediment propagule bank replenishment was lower than 25% for the South Lagoon, except in 2001 in the northern 30 km of the lagoon (Figure 39). This period corresponded with a sustained decline of *R. tuberosa* distribution, abundance and propagule bank and by 2011 plants were absent from all but the most northerly section of the South Lagoon and the propagule bank was depauperate (Paton 2001; Paton *et al.* 2001; Paton 2002; 2003; Nicol 2005; Paton 2005a; 2005b; Paton and Rogers 2008; Brookes *et al.* 2009; Whipp 2010).

The 2010-11 flood resulted in favourable hydrological conditions for *R. tuberosa* in the South Lagoon; therefore, higher probability of sediment propagule bank replenishment (Figure 39). Frahn *et al.* (2012) reported widespread but sparse occurrence of *R. tuberosa* in the South Lagoon between Parnka Point and Salt Creek

in December 2011. The abundance of *R. tuberosa* when sampled in December 2011 was much lower than reported in 1980s, 1990s and early 2000s (Frahn *et al.* 2012). This was not surprising given the unprecedented period of poor hydrological conditions in the ten years prior to sampling, which needs to be taken into consideration when interoperating future model outputs (i.e. high probability of sediment propagule bank replenishment may not result in widespread and abundant *R. tuberosa* in the South Lagoon). The high probability of propagule bank replenishment in 2013 also did not translate into abundant extant *R. tuberosa* or abundant propagule bank in 2014 (Frahn and Gehrig 2015). This suggests that an extended period of good hydrological conditions and interventions (i.e. importing propagules into the system) are required to re-establish abundant populations of *R. tuberosa* in the South Lagoon.

Model simulations for 2011 and 2013 suggested that there was little benefit in providing environmental water for the South Lagoon; however, there was a benefit to *R. tuberosa* populations from environmental water provided in 2012 (Figure 40). It is worth noting that the water provided in 2012 and 2013 was in addition to an unregulated flow and it is unlikely the volumes of environmental water available will provide any benefit to *R. tuberosa* in isolation. The years when there was a high probability of propagule bank replenishment corresponded with years when there was an unregulated flow of sufficient duration that resulted in barrage outflows during late spring and early summer. Nevertheless, the volumes of environmental water available could be delivered during late spring and early summer to maximise benefits for *R. tuberosa* by slowing the rate of declining water levels during this period.

Fish habitat

Salinities in the Coorong are highly variable and strongly driven by freshwater flows from the Murray River and tidal seawater exchange through the Murray Mouth (Geddes and Butler 1984; Joehnk *et al.* 2014). Typically, there is a strong north to south salinity gradient which influences the distribution, abundance and assemblage structure of fish species (Ye *et al.* 2012; Livore *et al.* 2013; Ye *et al.* 2015c). Fish habitat modelling using salinity tolerance threshold (LC_{10}) of key species provides a simplified preliminary assessment of the probability and extent of the suitable fish habitat in the Coorong subject to barrage flow releases with or without environmental water deliveries.

For periods where information is available on the distribution of particular species in the Coorong (the 1980s onwards) there is generally a strong correlation between the model outputs and findings from fish studies (Geddes and Butler 1984; Geddes 1987; Noell *et al.* 2009; Ye *et al.* 2012; Livore *et al.* 2013; Ye *et al.* 2015c) or commercial fishery catch data from the Coorong (SARDI Fisheries Statistics). For example, the reduction in modelled fish habitat due to increasing salinity, particularly in the southern part of the Coorong during drought periods, was consistent with results of several fish and habitat studies in the Coorong during the 1982 drought (Geddes and Butler 1984) and the more recent millennium drought from 2006–2009 (Noell *et al*. 2009; Ye *et al.* 2011a; Ferguson *et al.* 2013). On the other hand, the range and habitat extension for a number of fish species following the restoration of Murray River inflows to the Coorong was also demonstrated in results from several field studies in 1983-84 (Geddes 1987) and post 2010-11 flood (Livore *et al.* 2013; Ye *et al.* 2013c).

Results of the daily threshold analysis were consistent with the annual exceedance probability analysis. However, the habitat suitability shown in Figure 41 provides better resolution. While the threshold analysis may provide an indication of the impact of changes in the extent of a particular species' salinity threshold or habitat, it provides a binary (yes or no) output in contrast to the gradual habitat probability shown in Figure 41. Thus, daily threshold analysis lacks sufficient resolution to demonstrate intermediate regions of suitability. Annual habitat probability provides a more detailed picture summarising changes in salinity and thus habitat suitability over an annual cycle. For example, the *Ruppia tuberosa* response model may be extended in future to include updated life cycle information and thus deliver a further improved model of habitat suitability.

Evaluation of the effects of 2013-14 environmental watering on the annual habitat suitability suggested that only three out of the seven species, mulloway, Tamar goby and yelloweye mullet, benefited from environmental watering, i.e. scenarios 2 and 3, due to reduced salinity in the South Lagoon. For the other five species, the simulated salinity levels were below their respective thresholds throughout the Coorong in 2013-14 despite an annual barrage discharge of ~1,000 GL but following antecedent high to moderate flows since 2010-11 (Ye *et al.* 2015c). If similar amounts of environmental water was delivered during drought years, when most of the southern part of the Coorong became too saline for most fish species, it is likely that a greater

improvement of fish habitat would have occurred. Despite this, the simulation modelling for 2013-14 demonstrated a significant impact on habitat suitability for three fish species with lower salinity tolerances by withholding/delivering environmental water in the South Lagoon, where habitat suitability is generally much smaller than in the North Lagoon. For example, mulloway, as the most susceptible fish species in terms of salinity tolerance in this study, there would have been up to 25% reduction in habitat suitability and 40 km habitat contraction if no environmental water was delivered to the Coorong.

6 CONCLUSIONS AND LEARNINGS

6.1 Golden perch spawning and recruitment

Long-lived, native fish species require multiple age classes to increase population resilience during periods of unfavourable environmental conditions, including environmental perturbation and anthropogenic impacts. Consequently, assessment of population resilience requires an understanding of survivorship and population demographics. Recruitment of golden perch in the lower Murray River is facilitated by spawning that occurs in the lower Murray River and Darling River, and potentially also in the mid Murray River (Appendix: Figure A1). Spawning in the lower Murray River, in particular, has not previously been associated with regulated entitlement flows but instead with variability in flows (contained within-channel or overbank) in mid-late spring and throughout summer, generally at water temperatures >20 °C.

In 2013-14, the delivery of Commonwealth environmental water contributed to elevated flows (i.e. above *entitlement*) in the lower Murray River in late spring and throughout summer. Golden perch spawned in the lower Murray over this period, predominantly on the descending limb of a within-channel flow pulse, and low abundances of fish were recruited through to YOY in the floodplain and gorge geomorphic regions of the lower Murray River. Thus, the delivery of Commonwealth environmental water in 2013-14 supported the CEWO objective of providing a flow regime that supports breeding of native fish species and recruitment of juveniles.

6.2 Lateral movements of fish

A diverse fish assemblage was recorded moving between the main river channel and wetlands during the study period. Patterns of lateral fish movement were complex and highly variable among flow phases with marked differences between wetlands. Generally, movements of most species at each wetland, during each flow phase, were bidirectional and with no clear, consistent pattern relative to changes in flow conditions. Fish assemblages moving in and out of wetlands were influenced by inlet flow, flow in the main channel and water temperature. The magnitude of environmental flow in 2013-14 did not have noticeable effects on inlet flow and inlet

height of the two wetlands, therefore environmental water delivery likely had negligible influence on the lateral movement of fish assemblages.

6.3 Dissolved and particulate material transport

Modelled outputs from this study support the hypothesis that the flow regime, associated with Commonwealth environmental watering, increased the transport of dissolved and particulate matter through the Lower Murray River, Lower Lakes and Murray Mouth. Increased transport of this matter, associated with environmental watering, will play important functional roles in the studied ecosystems and the nearshore environment, providing habitat through reduced salinity levels and increasing productivity through the provision of resources. Salinity is a principal driver of habitat availability in the Lower Lakes and Coorong region, with elevated salinities severely associated with low inflows to the system severely compromising biotic populations (Brookes *et al*. 2009). Conversely, lower salinities resulting from increased inflows would be expected to increase habitat availability. Furthermore, the increased exports of nutrients and organic matter would likely increase secondary productivity, providing resources to support biotic populations within the region.

6.4 Coorong (modelling)

The operational hydrodynamic model (CHM v2.1), together with simple *Ruppia tuberosa* and fish habitat models, allow the assessment and evaluation of environmental water impacts on the ecological responses in the Coorong. In addition to hindcasting the effect of environmental watering, such a combined modelling system has the potential to provide future scenario/impact modelling of the effect of timing and quantity of environmental water on the ecosystem of the Coorong.

Ruppia tuberosa

The correlation between *R. tuberosa* distribution and abundance reported in the literature and the modelled probability of sediment propagule bank replenishment indicates that there is potential for the model to be used as a management tool. The model could be used to determine volume and timing of barrage outflows required to maintain viable populations of *R. tuberosa*. Furthermore, there is potential to use the model to investigate scenarios that may result in "false starts" (i.e. favourable

conditions for seed germination and turion sprouting followed by unfavourable conditions for life cycle completion).

Environmental watering in 2013-14 resulted in a significant increase in the chance of propagule bank replenishment for the North Lagoon of the Coorong compared to the model outputs without environmental water. Contrastingly, in the South Lagoon, environmental water delivery resulted in little difference to the probability of propagule bank replenishment.

Fish habitat

The preliminary fish habitat modelling based on salinity tolerance provides a simple tool to evaluate the habitat suitability and potential distribution of several key fish species in the Coorong. The model output was consistent with available field data. The model could be used for preliminary investigation to determine barrage outflow volumes required to maintain suitable habitat for key fish species along the Coorong during the cool and warm periods. Environmental watering in 2013-14 provided benefits for fish species in the Coorong by improving habitat suitability up to 25% and increasing habitat extent southward by up to 40 km. Nevertheless, understanding of the potential effects of environmental watering on fish populations in the Coorong provided by the current analysis is preliminary. Future fish response and habitat modelling could include life-history information for the different species/guilds which will provide a more comprehensive tool for evaluating and simulating environmental flow effects on population dynamics in the Coorong.

7 RECOMMENDATIONS

7.1 Golden perch spawning and recruitment

Environmental water management

In the MDB, seasonal increases in discharge promote the reproduction of flow-cued species, such as golden perch. Environmental water that contributes to flows of at least 15,000 ML day^{-1} from late spring through summer, when water temperatures exceed 20 °C, may promote the spawning and recruitment of golden perch in the Lower Murray River. Along with hydrological factors, the source and longitudinal continuity of the water may be integral to ecological outcomes.

Future monitoring/ research

The hydraulic characteristics of flow that may initiate golden perch spawning remain a priority research question in the MDB, along with the influence of flow source and longitudinal integrity. The river-scale population dynamics of golden perch also warrant further investigation, particularly spawning locations and the influence of juvenile and adult fish movement on population demographics.

7.2 Lateral movement of fish

Environmental water management

Flows in the main channel and wetland inlet explained most of the variability in bi-directional lateral movements of fish in Martin's Bend and Hart Lagoon wetlands. Although environmental flow in 2013-14 had no effect on wetland inlet flow, future environmental water deliveries of greater magnitude that influence inlet flow and inlet height are expected to support and enhance lateral fish movement in the Lower Murray River. These flows could be delivered in conjunction with natural unregulated flow events/pulses to maximise flow variability in wetland inlets. Nevertheless, integrated management (complementary actions) may be required to control invasive species (e.g. carp) to maximise outcomes for native fish species.

Future monitoring/research

Observations of the lateral movement patterns of native and invasive fish species in each wetland and how they were influenced by flow and other stimuli, provide an invaluable comparative database for future research and monitoring, building upon that of Ye *et al.* (2014). However, the high spatial variability in the structure of fish assemblages moving in and out of wetlands further demonstrates the need to further understand the effect of flow on lateral fish movement at other key wetlands in the Lower Murray River.

Findings from the current study suggest that Martin's Bend wetland and more specifically its inlet channel may provide suitable habitat for juvenile stages of large-bodied native fish species (e.g. golden perch). Future research should be focussed on understanding the role of these habitats for juvenile stages of large-bodied native fish species and gaining a more detailed understanding lateral moments of these species related to timing, duration and magnitude of flows. A better understanding of the lateral movements of native and invasive species will allow us to implement appropriate management strategies to dis-benefit invasive species, whilst at the same time having minimal impact on native species.

7.3 Dissolved and particulate material transport

Environmental water management

Based on insights provided by this study and previous studies, including Aldridge *et al*. (2013) and Ye *et al*. (2014), the following points may be used to help guide future environmental water use:

- environmental flow delivery can significantly influence salinity concentrations within the Murray Mouth region;
- environmental flow deliveries during extended low flow periods are likely to have greater impacts on concentrations of dissolved and particulate matter than periods with antecedent moderate flow conditions;
- environmental water use 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 day^{-1}) that inundate previously dry floodplain and wetland habitats;

- environmental watering during low to moderate flow periods (e.g. 10,000–40,000 ML day^{-1}) will increase the transport and export of dissolved and particulate matter;
- maximum exports of dissolved and particulate matter from the Murray Mouth are likely to be achieved by delivering environmental water during periods of low oceanic water levels (summer). However, this may reduce water availability at other times, increasing the import of matter from the Southern Ocean during those times. In contrast, delivery of environmental water to the Lower Murray River 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;
- net export of dissolved and particulate matter can be achieved when discharges above threshold levels are provided. Whilst, these discharge thresholds are currently unknown and likely differ with seasonal changes in downstream water levels, supplementary water sources are important in providing adequate flows to export matter from the system;
- flows during winter may result in limited assimilation of nutrients by biota (lower productivity related to lower temperature), whilst deliveries during summer could increase the risk of blackwater events and cyanobacterial blooms, depending on hydrological conditions. 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);
- achieving multiple ecological objectives may require multiple watering events in a given year. For example one event in spring could be provided to increase nutrient assimilation, followed by a subsequent event to export matter to downstream ecosystems.

However, a broader assessment of various scenarios (i.e. "what if" scenarios) is required in order to reach more general recommendations about optimal use of environmental water for the transport of dissolved and particulate matter because the Lower Murray is a hydrologically complex system.

Future monitoring/research

The approach used for this study was valuable for evaluating changes in concentrations and transport of dissolved and particulate matter associated with environmental water deliveries. Continued refinement of the model used in this study will further improve its capacity to evaluate the influence of environmental water delivery on dissolved and particulate matter concentrations and transport in the Lower Murray. This is dependent on the availability of data for validating the model. Sampling effort has decreased since 2011-12, but it is unclear whether water quality monitoring within the Lower Lakes and Coorong will continue (SARDI 2014). To ensure further insights can be provided through the application of this approach in future years, it is recommended that the existing water quality monitoring program is maintained or established through the long-term intervention monitoring project (SARDI 2014).

In the future, modelling approaches such as that used in this study, could be used to assess the potential benefits of various watering actions in both planning and reporting of ecological outcomes to environmental water deliveries. For reporting, additional "what if" scenarios could be tested to investigate what could have been achieved with alternative water actions. For planning, the relationship between season and discharge thresholds could be investigated for the purposes of setting flow targets for the delivery of environmental water. This could include information generated from a current Goyder Institute research project investigating food-web responses in the Coorong associated with nutrient pulses generated by providing flows across the barrages. Combined with outputs from this model used here, this information could be used to assess the likely benefits of resource inputs to secondary productivity (and the ecosystems more generally) under various flow scenarios.

7.4 Coorong (modelling)

Ruppia tuberosa

Environmental water management

The volumes of environmental water under current allocations will have little (if any) benefit on *R. tuberosa* populations in the South Lagoon unless delivered in conjunction with an unregulated flow event. Even in this situation, the unregulated flow will need to be of sufficient duration to provide barrage outflows during November and early December and environmental water used to manage flow recession to reduce the rate of water level decline in the South Lagoon to reduce the risk of stranding.

Future monitoring/research

The current *R. tuberosa* response model together with the operational hydrodynamic model (CHM v2.1) provides a useful tool for modelling alternate scenarios and evaluating them to assess the effect of environmental watering on *R. tuberosa*. Additional information regarding the salinity thresholds for flowering and seed production will improve the ecological response model and the environmental triggers for flowering in this species and inform environmental water management. Continued monitoring of the propagule bank and extant population of *R. tuberosa* in the Coorong will improve calibration of the model.

Research into the effects of sub-lethal salinity of submergent plant species that were historically present in the Coorong (e.g. *Ruppia megacarpa, Lamprothamnium macropogon, Lepilaena cylindrocarpa*) could result in similar models being developed for these species to evaluate the potential for their reintroduction. Furthermore, information regarding the salinity and water level preferences of fringing species (samphire and salt marsh species) could be used to develop ecological response models to assist in the management of fringing habitats.

Fish habitat

Environmental water management

Freshwater inflow is essential for maintaining estuarine fish habitat and populations in the Coorong. Environmental water delivery has the potential to provide incremental benefits by reducing salinity levels along the north to south gradient in the Coorong, resulting in extension of habitat into the South Lagoon, particularly for species with relatively low salinity tolerance. Environmental watering during the summer months, or in years with relatively low barrage flows and higher salinity, will provide a much larger effect of habitat change. Flow delivery during late spring/summer is important as this period corresponds to the spawning and recruitment season of most estuarine fish species in the Coorong. Environmental flows could potentially help in maintaining a favourable salinity gradient, enhancing productivity and improving connectivity to facilitate fish recruitment.

Future monitoring/research

The preliminary fish habitat model together with the operational hydrodynamic model (CHM v2.1) provides a useful tool for broad evaluation and scenario modelling to assess the effect of environmental watering on fish habitat suitability in the Coorong. In order for more comprehensive analyses and evaluation of the effect of environmental watering on fish habitat and populations, more complex fish models are required. This could be achieved by extending the existing fish habitat model with information on life-histories of key species and the effects of flow/salinity on key life-history stages and processes across species/guilds. Further investigation and data collection could be achieved through ecological monitoring of the effects of flow, including environmental water delivery, on fish assemblages, habitat availability and recruitment of key fish species in the Coorong. Such knowledge will not only inform the future development of fish–flow response models but also contribute to understanding of flow related ecology to underpin environmental water management for maintenance and improvement ecosystem health and resilience in the Coorong.

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9 APPENDICES

I. Golden perch spawning and recruitment

Southern MDB sampling sites for water 87Sr/86Sr

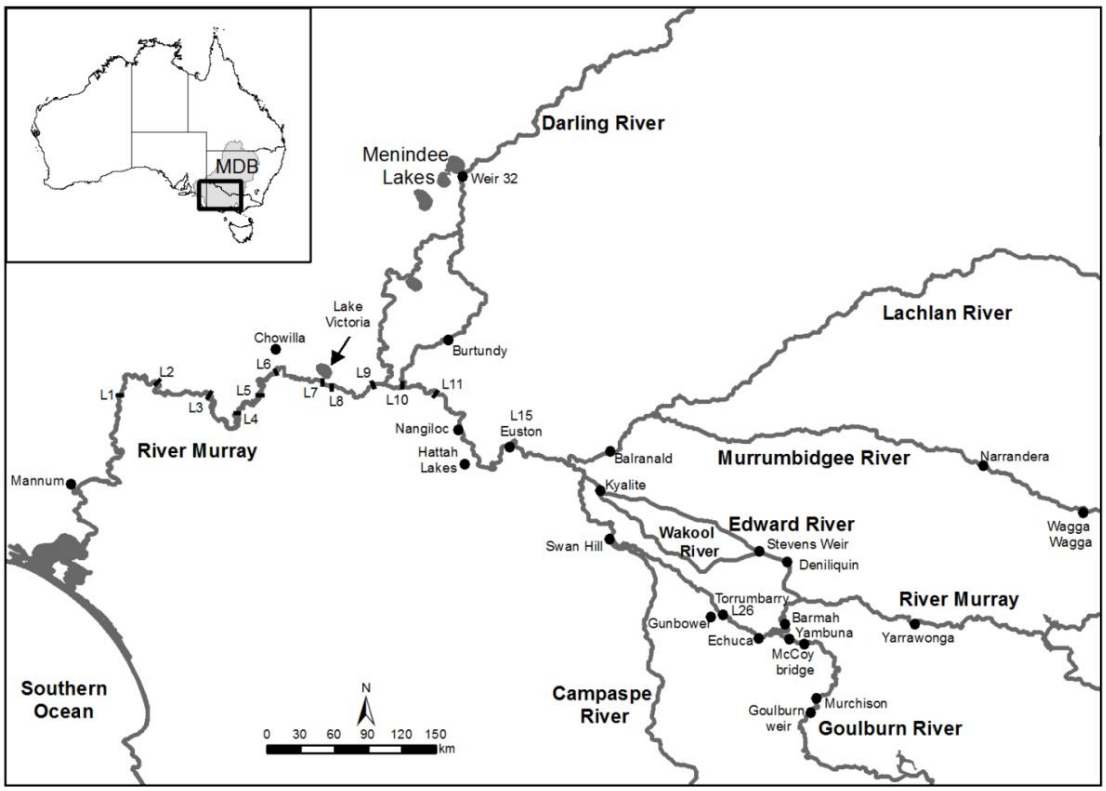

Figure A1 - Map showing the location of the Murray–Darling Basin and the major rivers that comprise the southern Murray-Darling Basin, showing the numbered Locks and Weirs (up to Lock 26, Torrumbarry), the Darling, Lachlan, Murrumbidgee, Edward–Wakool, Campaspe and Goulburn rivers and Lake Victoria, an off-stream storage used to regulate flows in the lower Murray River.

II. Dissolved and particulate material transport

Sampling sites within each water-body

Water-body	Sampling site	X	Y	Zone	Sampling dates	Data source
Lower Murray River	Blanchetown (Lock 1)	373768.9	6195837	UTM 54S	Weekly between 04/07/2013 and 26/06/14	SA Water
	Wellington	353152.5	6086654	UTM 54S	24/07/2013, 03/10/2013, 24/10/2013, 22/11/2013, 18/12/2013, 21/01/2014, 16/04/2014	Murray Futures (DEWNR/EPA)
Lower Lakes	Lake Albert Middle	346724.2	6056213	UTM 54S		Murray Futures (DEWNR/EPA)
	Lake Alexandrina Opening	348468.4	6080369	UTM 54S		
	Lake Alexandrina Middle	331761.4	6077836	UTM 54S		
Coorong	Point McLeay	324379.4	6068672	UTM 54S	24/09/2013, 22/10/2013, 21/11/2013, 17/12/2013, 22/01/2014	Murray Futures (DEWNR/EPA)
	Goolwa Barrage	302100.7	6066302	UTM 54S		
	Murray Mouth	308001.3	6063098	UTM 54S		
	Ewe Island	315228.9	6062110	UTM 54S		
	Mark Point	325761.5	6054914	UTM 54S		
	Parnka Point	355250.6	6025735	UTM 54S		

Model set-up

To assess the effects of the environmental water provisions on salt and nutrient loads moving through the region below Lock 1 a single model domain was applied spanning Lock 1 to the Southern Ocean, including the North Coorong east to Parnka Point (Figure A2). The model platform used was the coupled hydrodynamic-biogeochemical model TUFLOW-FV-AED, developed by BMTWBM and the University of Western Australia. The TUFLOW-FV model (BMTWBM 2014) adopts an unstructured-grid model that simulates velocity, temperature and salinity dynamics in response to meteorological and inflow dynamics. In this application, AED was configured to simulate the dynamics of light, oxygen, nutrients, organic matter, turbidity and phytoplankton (Hipsey *et al*. 2013). The model approach adopted within the AED model was conceptually similar to earlier studies (Hipsey and Busch 2012; Aldridge *et al*. 2013) that adopted the CAEDYM model platform.

The model runs were initialised with data from a range of data sources. For this study, four simulations were run from July 2013 to June 2014, including three environmental watering scenarios and base-case without environmental water (Figure A3). Inflow data (Lock 1) used to drive the main river domain were provided by the MDBA for the scenarios. Water quality conditions for the river inflow at Lock 1 were determined based on interpolation of available data from Lock 1 or Morgan. For water quality properties for the base-case (without environmental water provision) scenario, rating curves were developed for flow and concentration. Based on the daily flow difference, a scaled concentration was estimated for water quality parameters including salinity, phosphate, ammonium, nitrate, total nitrogen and silica. Additional flow specifications for SA Water off-takes were also included. Irrigation return flows were assumed to be negligible over this period and were not included in the model. Similarly, flows from Eastern Mount Lofty Ranges were not included since their contribution to the Lower Lakes during periods of high Murray River inflows is minor (Cook *et al.* 2010). Meteorological conditions were based on data from Narrung.

Between Lake Alexandrina and the Coorong four barrages were included (Goolwa, Mundoo, Ewe Island and Tauwitchere) and set with a spill-over height of 0.72 mAHD. The barrage operation was set to include gate operation based on operational information provided through discussions with modellers within the Department of

Environment, Water and Natural Resources. At the bottom of the domain, two open boundaries were specified, one at the Murray Mouth and one at Parnka Point. Murray Mouth water level was based on Victor Harbor tidal data, which is available at 10 min resolution. Parnka Point height data was set based on available water level data from the WaterConnect online data reporting site. Water quality conditions for both boundary points were set based on a linear interpolation of the measured nutrient and salinity data collected as part of this study.

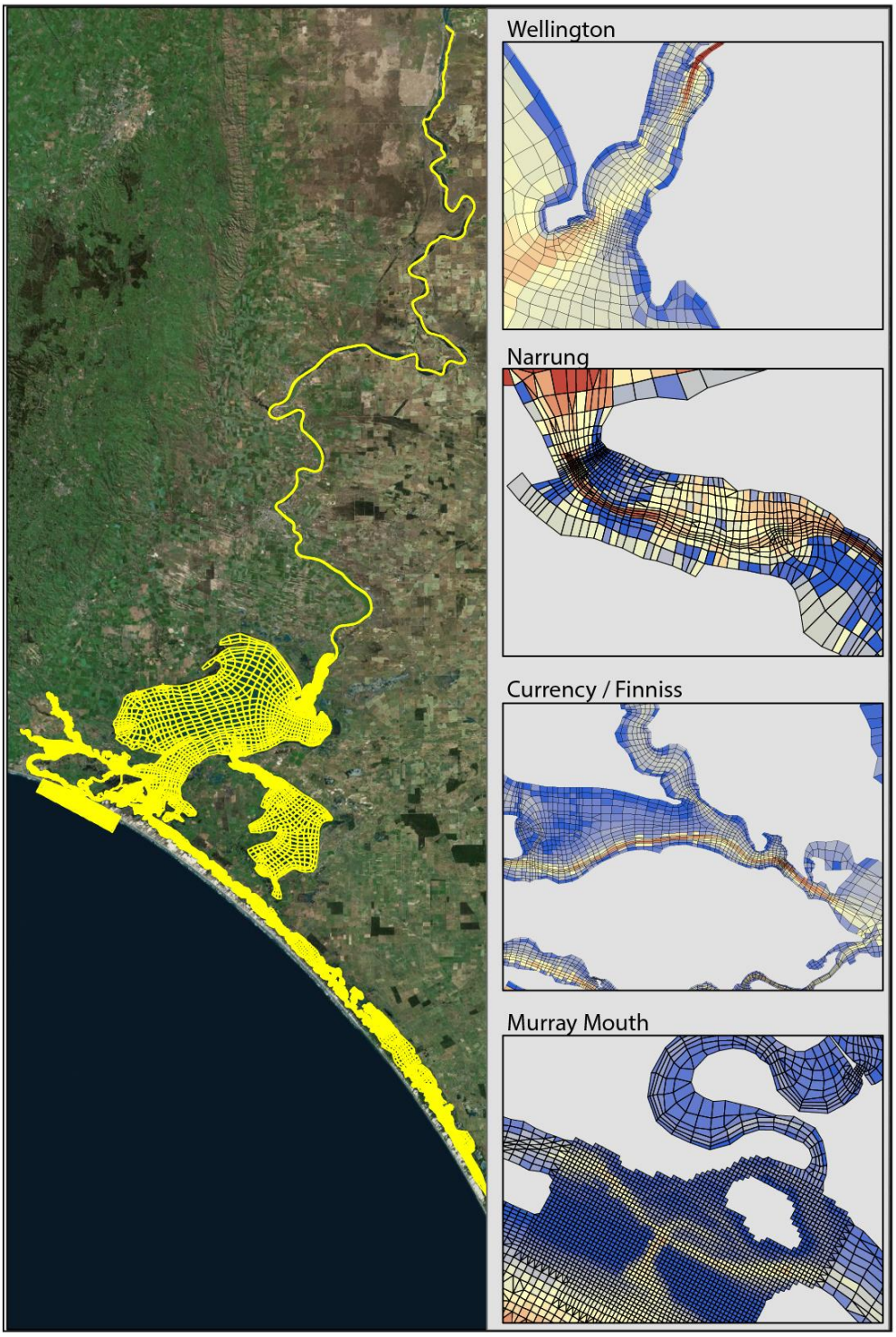

Figure A2 - Overview of model domain applied in this study using TUFLOW-FV. Grid provided courtesy of Department of Environment, Water and Natural Resources.

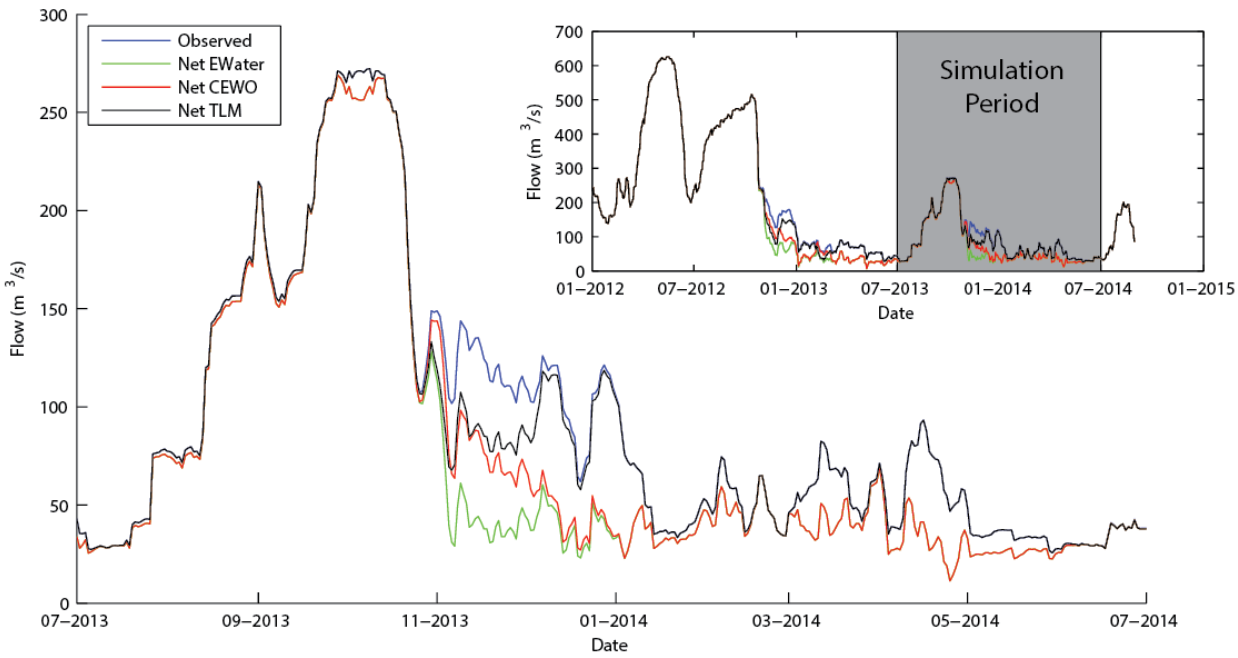

Figure A3. Overview of four flow scenarios assessed by the model simulations. Flows were applied to the model at the upstream Lock1 boundary.

Acronyms

AHD	Australian Height Datum
CEW	Commonwealth environmental water
CEWO	Commonwealth Environmental Water Holder
CSIRO	Commonwealth Scientific and Industrial Research Organisation
DEWNR	Department of Environment, Water and Natural Resources
EPA (org.)	Environment Protection Authority
EW	Environmental water
FRM	Flood recruitment model
LK-1	Lock 1
LK-6	Lock 6
LK-6A	Lock 6 anabranch
LMR	Lower Murray River
LFRH	Low-flow recruitment model
MDB	Murray-Darling Basin
NRMB	Natural Resources Management Board
Rim-FIM	River Murray Flood Inundation Model
SAMDB	South Australian Murray-Darling Basin
SARDI	South Australian Research and Development Institute
TL	Total length
TLM	The Living Murray
TLMW	The Living Murray Water
VEWH	Victorian Environmental Water Holder
YOY	Young-of-year