Inland Waters and Catchment Ecology

Spatial and temporal variations in larval fish assemblages between Locks 1 and 6 in the River Murray, South Australia: with reference to drought intervention monitoring 2007.

SARDI Aquatic Sciences Publication Number: F2008/000685-1
SARDI Research Report Series Number: 304

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2008

Government of South Australia
South Australian Murray-Darling Basin
Natural Resources Management Board

Spatial and temporal variations in larval fish assemblages between Locks 1 and 6 in the River Murray, South Australia: with reference to drought intervention monitoring 2007.

A report to the SA MDBC NRM board under the project title:
Effect of weir pool lowering below Lock 1 including the Lower Lakes [Part 2]:
Drought resistance investigations for spawning and recruitment of native fish species in the lower River Murray SA.

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2008

This Publication may be cited as:

Cheshire, K. and Ye, Q. (2008) Spatial and temporal variations in larval fish assemblages between Locks 1 and 6 in the River Murray, South Australia: with reference to drought intervention monitoring 2007. SARDI research report series number: 304, SARDI Aquatic Sciences Publication Number F2008/000685-1, SARDI Aquatic Sciences, Adelaide, pp 41.

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SARDI Research Report Series Number: 304
SARDI Aquatic Sciences Publication Number: F2008/000685-1

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Signed:
Date: 6 February 2009
Distribution: SA Murray-Darling Basin Natural Resource Management Board
Circulation: Public Domain

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ACKNOWLEDGEMENTS

Funding from the Murray-Darling Basin Commission's The Living Murray Intervention Monitoring Program, provided through the South Australian Murray-Darling Basin Natural Resources Management (SA MDBC NRM) Board supported this research.

Lesley Alton, Phillipa Wilson and Luciana Buccater were integral in assisting KC with sorting and identifying the 1000's of fish collected throughout the duration of this project. Special thanks to Michael Guderian and Matt Pellizzari for the major roles taken as field staff and lab sorters. We also thank Neil Wellman, Michelle Roberts, Natalie Bool, Catherine Lawless, David Short, and David Fleer for field and lab assistance.

DWLBC Knowledge and Information Division provided discharge, water level, temperature and conductivity data for the sites throughout the Lower River Murray, thanks to Barry Porter for advice with using this information.

Thank you to Jan Whittle, Rohan Henry and Adrienne Frears for the roles taken as project managers through the SA MDBC NRM board. Similarly, thank you to David Hohnberg and Stuart Little for their roles taken through the MDBC, and for conducting the MDBC review and providing constructive comments on the final draft of the manuscript.

Thank you to Kelly Marsland and Brenton Zampatti for conducting the SARDI internal review on the draft report and providing constructive advice to improve the manuscript. Thankyou to Suzanne Bennett for organising the SARDI review and publication process.

EXECUTIVE SUMMARY

The Murray-Darling Basin (MDB) in south eastern Australia is currently experiencing one of the most severe hydrological droughts on record. Prolonged periods of low flow have the potential to severely impact aquatic ecosystems. Water quality may decline, and low water levels and the absence of hydrological cues may inhibit recruitment of some invertebrates, fish and plant species. Native fish populations have declined in abundance and range, following changes to the natural flow regimes as a result of river regulation. When combining the effects of river regulation with the severe drought that began in 2002, the impact on the already stressed native fish populations is likely to be intensified. Therefore, to conserve native fish, it is important to understand the biological responses as well as the life history patterns and survivorship strategies of key species during the current critical conditions.

Larval fish sampling within the main channel of the Lower River Murray, South Australia (SA) was conducted during the spring/summer in 2005, 2006 and 2007, at selected sites downstream of Locks 1 and 6. Sampling was conducted fortnightly from October to December, using plankton tows in pelagic environments, in an effort to determine if change in inter-annual and spatial patterns could be detected for key native species. Discharge volume, water level, temperature, and conductivity, were also monitored to determine if correlations between the larval fish assemblage and any environmental variables could be identified.

Increased discharge was experienced in the river in 2005, compared with below entitlement discharge in 2006 and 2007. During 2005, larval fish comprising 11 species, nine native and two exotic, were collected at all sites, including the large-bodied native species, Murray cod, golden perch, silver perch, and freshwater catfish. In 2006 and 2007, larvae of nine species were collected, consisting of seven native and two exotic, and notably, the larvae of two large-bodied native species, golden perch and silver perch, were absent. There were significant inter-annual and spatial differences in assemblage structure of the small-bodied native species during the study. For the large-bodied native species, the assemblage structure was significantly different in 2005 from 2006 and 2007, but there was no difference between 2006 and 2007.

Overall inter-annual variation in hydrology seems to have been the greatest contributing factor shaping the inter-annual variation in both small-bodied and large-bodied larval assemblages. There were two distinct responses identified for the larval assemblages in relation to differing discharge volume. The small-bodied native species responded positively, increasing in larval abundance during the lower discharge years, 2006 and 2007. However, for the large-bodied native species, the abundance and diversity of larval fish was the highest in 2005, although the number of larvae collected was very low throughout the study. The absence of golden perch and silver perch larvae in 2006 and 2007 indicates no spawning success in the Lower Murray River channel for these species during years of lower discharge. In addition, there is evidence in the literature to

suggest that Murray cod larval survivorship and recruitment may be impacted by low flow conditions. Downstream of Lock 1 appears most affected by the current drought conditions, of particular concern is the decrease in bony herring larvae in 2007 and the absence of large-bodied native species in both 2006 and 2007.

This study suggests protracted low flow conditions pose a risk to spawning success and larval survivorship of freshwater catfish, Murray cod, golden perch and silver perch. Given their significant conservation value in the MDB, these species should be of greatest concern. This study has shown that in SA, under current low flow conditions, diversity, abundance and distribution of the larvae of large-bodied native species is decreased. Furthermore, a within channel flow pulse may be sufficient to provide suitable conditions for spawning and enhanced larval survivorship of the large-bodied native species. The large-bodied native species are long lived, and therefore, populations are likely to be able to withstand unfavourable conditions over the short term. However, if current conditions continue, without effective long-term management we may face significant loss of some of our most valuable species. Water management strategies, in the form of environmental water allocations, need to be developed to ensure that during periods of continued low flow, water is available to allow for management of flow pulses, thus, providing contingency plans for large-bodied native fish populations. Discussion is made at the end of this document on management suggestions and further research that is required to gain a better understanding of the dynamics of the system.

1 INTRODUCTION

The Murray-Darling Basin (MDB) in south eastern Australia is currently experiencing one of the most severe hydrological droughts on record (MDBC 2007; Murphy and Timbal 2008). Inflows into the MDB during 2006/07 were only 55 % of the previous minimum recorded (MDBC 2007); in addition temperatures have been above average adding to the impact of low flows (MDBC 2007; Murphy and Timbal 2008). Prolonged periods of low flow, such as are currently occurring, have the potential to severely impact aquatic ecosystems. Water quality may decline, and low water levels and the absence of hydrological cues may inhibit recruitment of some invertebrates, fish and plant species (Bunn and Arthington 2002).

The aquatic ecosystems of the MDB have been severely affected by anthropogenic influences, particularly river regulation and water abstraction schemes (Walker 1985; Walker and Thoms 1993; Arthington and Pusey 2003). River regulation impacts on natural flow variability by reducing the frequency and duration of major flooding events and the magnitude of smaller within channel flows, and by maintaining a relatively stable water level within the weir pools (Maheshwari *et al.* 1995). Native fish populations have declined in abundance and range, following changes to the natural flow regimes (Cadawallader 1977; Gehrke *et al.* 1995; Humphries *et al.* 2002; MDBC 2004). Aspects of river regulation can impact on fish through affecting necessary conditions for spawning (e.g. cues for gonad maturation, migration and pre-spawning interactions) and for recruitment (e.g. decoupling the occurrence of larvae and conditions required for survival through to juveniles) (Humphries and Lake 2000). Knowledge of the role of flows and flooding in the life history cycles of many of the MDB fish have improved in recent years (Humphries *et al.* 1999; CRCFE 2003; King *et al.* 2003; Cheshire and Ye 2008; Leigh *et al.* 2008). However, the spawning responses of native fish populations during drought conditions are poorly understood in Australia. When combining the effects of river regulation with the severe drought that began in 2002, the impact on the already stressed native fish populations is likely to be intensified. Therefore, it is important to understand the biological responses as well as the life history patterns and survivorship strategies of key species during critical conditions, in an effort to improve knowledge for native fish conservation and manage these species accordingly.

The life cycle strategies employed by native fish can be broadly divided into three categories (after Humphries *et al.* 1999), Mode 1, 2 and 3. Murray cod (*Maccullochella peelii peelii*) and freshwater catfish (*Tandanus tandanus*), are considered Mode 1 spawners, while not requiring an increase in flow to initiate spawning, they receive significant benefits during periods of increased flow, which improve larval survivorship (Humphries *et al.* 1999; Ye *et al.* 2000; Koehn and Harrington 2006; Ye and Zampatti 2007; Cheshire and Ye 2008; Leigh *et al.* 2008). Mode 2 spawners, golden perch (*Macquaria ambigua*) and silver perch (*Bidyanus bidyanus*), are considered flood/flow cued spawners, that require increases in the discharge/flow to initiate spawning and promote larval survivorship (Humphries *et al.* 1999; Mallen-Cooper and Stuart 2003; Gilligan and Schiller 2004; King *et*

al. 2005; Ye 2005; King *et al.* 2007; Cheshire and Ye 2008; Leigh *et al.* 2008). The small-bodied native species, (Australian smelt (*Retropinna semoni*), carp gudgeons (*Hypseleotris* sp.), flathead gudgeons (*Philypnodon grandiceps*), bony herring (*Nematalosa erebi*) and hardyheads (*Craterocephalus* sp.)), fall under the Mode 3a and b classifications and have a diverse range of spawning strategies, these species utilise a diverse range of habitats for spawning and recruitment, to enhance spawning during low flow years (Humphries *et al.* 1999; King 2002; CRCFE 2003; Cheshire and Ye 2008).

The current discharge into the Lower River Murray, as measured at the South Australian (SA) border, is the lowest on record since river regulation began in the 1930's, and the low flow pattern is predicted to continue (MDBC 2008). Furthermore, as a consequence of ongoing low flows, weir pool levels in the Lower River Murray are well below those normally maintained, this is particularly pronounced downstream of Lock 1 (DWLBC 2008). The reduced water levels have significant implications for a number of native fish species in the river, through affecting conditions necessary for spawning and/or larval survivorship. This report details the inter-annual and spatial changes in larval assemblages in the main channel of the Lower River Murray, in 2005 (above entitlement flow year), 2006 (below entitlement flow year) and 2007 (below entitlement flow year) to investigate whether drought conditions have had a further impact on the spawning and survivorship of native fish larvae.

2 OUTCOMES AND OBJECTIVES

The low volume of water in the Lower River Murray, as a consequence of the current drought conditions, may have an impact on the spawning success of native fish. It is important to understand the biological responses/performances of key species during the current critical conditions, to allow informed management decisions in the future. Studying larval fish assemblages and comparing inter-annual differences is an efficient way in which to provide an insight into the spawning success of the key fish species.

The main aims of this project were to:

- Describe the inter-annual variation for larval fish assemblage structure between 2005, 2006 and 2007, in relation to environmental variables in the River Murray, SA.
- Determine if spatial variation in the larval fish assemblage is occurring between sites downstream of Locks 1 and 6 during 2007, in relation to differences in the environmental variables, in the Lower River Murray, SA.

3 MATERIALS AND METHODS

3.1 Study sites

The Murray-Darling is Australia's largest river catchment, occurring in south-eastern Australia between the latitudes of 24-37 ° S and longitudes 138 – 153 ° E. It stretches 2,560 km, and covers an area of 1,063,000 km^2 (Newman 2000). The rainfall throughout most of the basin is low, and evaporation rates are high, as most of the basin is in semi-arid to arid climatic regions. The present study was confined to the main channel of the River Murray in SA, in both the gorge and floodplain regions (Figure 1). The South Australian section of the Murray is a heavily regulated lowland temperate river, most stream diversity occurs during high flows when adjacent floodplains become inundated. In SA, the River Murray encompasses two distinct geo-morphological zones, "floodplain" and "gorge", with different ecological features. The floodplain region, surrounding Lock 6, is a wide floodplain (5-10 km), comprising a variety of aquatic habitats including pools, anabranch channels, billabongs, wetlands, and the main channel is characterized by large woody debris (LWD) and macrophytes (Walker and Thoms 1993; Young 2001). The lowland gorge, surrounding Lock 1, is an incised section characterised by limestone cliffs, the floodplain is constrained (2-3 km), the main channel is dominated by pool habitats, and off-channel habitats are mostly wetlands (Walker and Thoms 1993; Young 2001). Sampling was conducted at two sites, the gorge section downstream of Lock 1 (34°21.138' S, 139°37.061' E), and the floodplain section downstream of Lock 6 (33°59.725' S, 140°53.152) (Figure 1).

3.2 Sampling trips

Larval fish sampling was conducted during the spring/summer of 2005, 2006 and 2007 from October through December. This sampling period was selected based on the peak spawning season and larval abundances within the river system as identified by Cheshire and Ye (2008), Humphries *et al* (2002), Leigh *et al.* (2008) and Meredith *et al.* (2002). Sampling was conducted fortnightly within each season resulting in six trips per year. Each site was sampled during the day and at night, of the same day, and all sites were sampled within a three-day period.

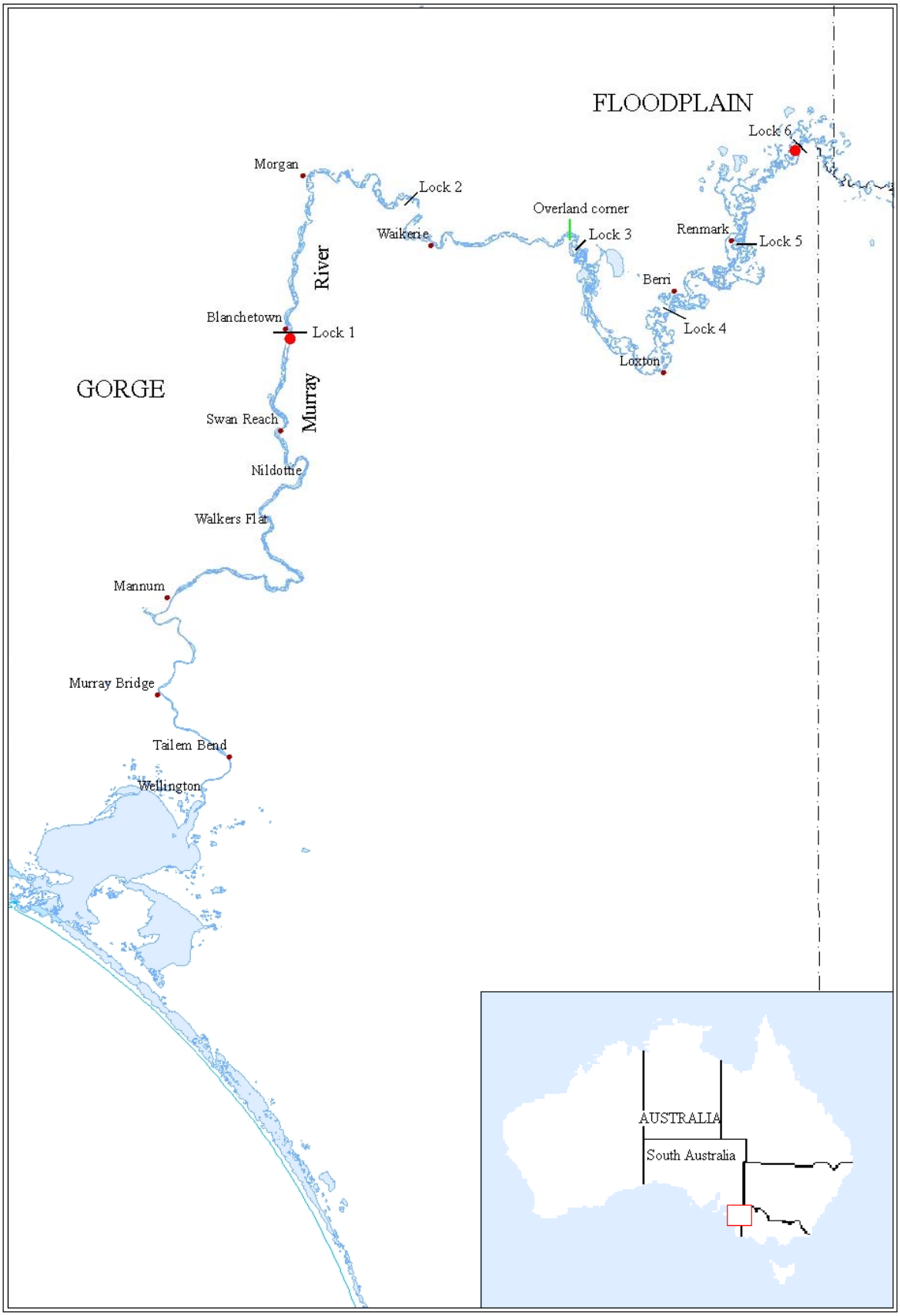

Figure 1. A map of the River Murray and Lower Lakes in South Australia, showing the six locks. The larval fish sampling sites below Locks 1 and 6 are highlighted by the red circles.

3.3 Sampling equipment and methodology

Plankton tows were conducted using a pair of square-framed, 0.5 x 0.5 m, 3 m long bongo nets of 500 µm mesh. Nets were equipped with 30 cm pneumatic floats either side of the frame, so the frame sat 5 cm below the water surface. They were towed in circles using a 20 m rope, in the centre of the main river channel. Four 15-min tows were conducted below each lock during the day and again at night giving a total of eight tows below each lock (four day and four night). Samples from left and right nets were grouped for analysis. The volume of water filtered through each net was determined using a flow meter (General Oceanics), fitted in the centre of the mouth openings. Plankton tow data were standardised to number of larvae per 1,000 m^3.

Figure 2. Bongo nets used for plankton tows

3.4 Sorting and identification

Samples were immediately preserved in 95% ethanol *in situ* and returned to the laboratory for sorting using magnification lamps. All larvae were identified to species level, where possible, using published descriptions (Lake 1967; Puckridge and Walker 1990; Neira *et al.* 1998; Serafini and Humphries 2004), with the exception of carp gudgeons (*Hypseleotris* sp.) and hardyheads (*Craterocephalus* sp.). Each of these two genera were treated as species complex due to close phylogenetic relationships and very similar morphologies (Serafini and Humphries 2004). Since large-bodied fish species such as Murray cod, golden perch, silver perch and freshwater catfish, can be difficult to collect as true larvae, immediately post-larval fish were included in counts; juvenile fish were noted but not included in the analysis.

3.5 Measurement of environmental variables

On each sampling trip in situ measurements (using a TDS water quality meter) were taken at 0.2 m and 3 m below the surface, during the day and at night for dissolved oxygen (mg/L), and pH. Turbidity (depth mm) was also determined using a secchi disc during the day at each site for each sampling event. Data for discharge (ML per day), water level (mAHD[1]), water temperature (°C) and conductivity (µS/cm @ 25 °C) were obtained for all sites, throughout the entire season, from the Department for Water, Land and Biodiversity Conservation, Knowledge and Information Division, Surface Water Archive. (http://e-nrims.dwlbc.sa.gov.au/swa/)

3.6 Data analysis

Plankton tow data were standardised to number of larvae per 1,000 m^3, and left and right plankton tow nets were grouped for analysis. This resulted in 288 samples (3 years x 6 trips x 2 sites x 8 replicates). The average of the replicates for each trip, at each site was calculated for graphical representation for individual species. Given knowledge of life history characteristics, including spawning, body size, age at maturity, and life span, the species collected were separated into three groups 1) small-bodied native species (Australian smelt, carp gudgeons, flathead gudgeons and bony herring[2]), 2) large-bodied native species (Murray cod, freshwater catfish, golden perch and silver perch) and 3) exotic species (common carp (*Cyprinus carpio*) and redfin perch (*Perca fluviatilis*)). The small-bodied native species and the large-bodied native species were analysed separately to determine results in line with spawning strategies. The sampling period did not cover the main spawning season for the exotic species, therefore no further statistical analyses were performed for this group.

Multivariate statistical analyses were performed following the methods described by Clarke (1993) using PRIMER v.6 (Plymouth Routines in Multivariate Ecological Research) software package. Prior to analysis the small-bodied native species data were fourth-root transformed to prevent highly abundant species from influencing the similarity measure (Clarke 1993). A dummy variable ("No species" with an abundance of one) was added to the large-bodied native species assemblage to allow for the analysis of samples with no species present. Bray-Curtis similarity measures (Bray and Curtis 1957) were used for all analyses. As the data did not satisfy assumptions of normality and homogeneity of variance, permutational multivariate analysis of variance (PERMANOVA: Anderson 2001) was used to detect any differences between year and site in both small-bodied native and large-bodied native assemblages. In both analyses year and site were treated as fixed factors. Unrestricted permutation of the data was used, with 999 permutations, to detect differences at α=0.05 (Anderson 2001). Where a low number of unique permutations occurred, the Monte-Carlo p –values

[1] AHD = Level relative to Australian Height Datum
[2] Bony herring is usually qualified as a medium sized fish- however the larvae are very similar to those of the small bodied species so for the context of this report they have been included as a small-bodied species.

(p MC) were included and used to test for significant differences (Anderson 2005). If significant differences were detected, pairwise tests were subsequently conducted to determine where the difference occurred. A similarity percentages (SIMPER) analysis was also performed to identify species contributing most to the differences between groups no cut-off level was applied. Non-metric Multidimensional Scaling (MDS) ordination was used to visualise the temporal and/or spatial patterns.

3.6.1 Linking larval fish assemblages with environmental variables

To determine whether environmental variables influenced small-bodied and large bodied larval assemblages daily values for the environmental variables were selected to generate a set of four replicates for each site and trip. The routine BIOENV (Clarke and Warwick 2001) was used for this correlation analysis for within site inter-annual variation, permutational analysis (conducting 999 permutations) was also conducted to provide a significance value for the correlations. The rank similarity matrices (Bray-Curtis similarity for fish assemblages and normalised Euclidean distance for environmental variables) were compared using the Spearman rank correlation. The rank correlation coefficient (ρ_s) lies between -1 and +1, corresponding to the cases where the fish assemblage and environmental patterns are in complete opposition or complete agreement (Clarke and Ainsworth 1993). Where distinct drivers were identified, circles scaled in size to represent the values of the 'best fit' environmental variables were also individually superimposed onto the two-dimensional MDS plots of both small-bodied and large-bodied native fish assemblages in order to identify visual concordance (Field *et al.* 1982).

4 RESULTS

4.1 Environmental variables

Discharge at both Locks 1 and 6 was higher in spring/summer of 2005 than during the same time period in 2006 or 2007 (Figure 3a). Throughout 2005 discharge exceeded the summer entitlement allocation of ~ 7, 000 ML per day. The peak discharge was reached during November 2005 with a maximum of 13,700 and 12,200 ML per day at Locks 1 and 6, respectively (Figure 3a). Discharge in 2006 was generally below entitlement allocation, due to drought conditions (Figure 3a). During 2007, discharge remained well below the summer entitlement allocation during sampling, reaching maximums of 3,040 and 2,360 ML per day, downstream of Locks 1 and 6, respectively. Discharge was similar within years between both Locks with only minor differences existing, as a result of geographical placement (Figure 3a). Water is diverted around Lock 6 through the Chowilla anabranch, accounting for the lower discharge values, when compared with Lock 1.

Water level differed between years and sites, and was consistently higher at Lock 6 than Lock 1 (Figure 3b). Water level downstream of Lock 6 was highest in 2005, and remained relatively stable in 2006 and 2007 (Figure 3b). Downstream of Lock 1 water level remained relatively consistent during 2005 and 2006, however, water level was lower during 2007 (Figure 3b).

Mean daily water temperature was consistent between sites, and increased steadily from October to December, with similar patterns in all years (Figure 3c). Temperature increased from approximately 16 °C in late September peaked in late January at approximately 28 °C (Figure 3c).

There was spatial and inter-annual variation in conductivity. Conductivity downstream of Lock 1 was consistently higher than downstream of Lock 6 (Figure 3d). Downstream of Lock 1 conductivity increased in each consecutive sampling year, ranging from 244-370 µS/cm, 371-455 µS/cm and 499-770 µS/cm, during 2005, 2006 and 2007, respectively (Figure 3d). Conductivity downstream of Lock 6 was consistent during 2005 and 2006 ranging from 140-205 µS/cm in both years, but was higher in 2007, ranging from 187-295 µS/cm (Figure 3d). Additionally, during 2007 conductivity peaked prior to sampling and steadily decreased at both sites (Figure 3d).

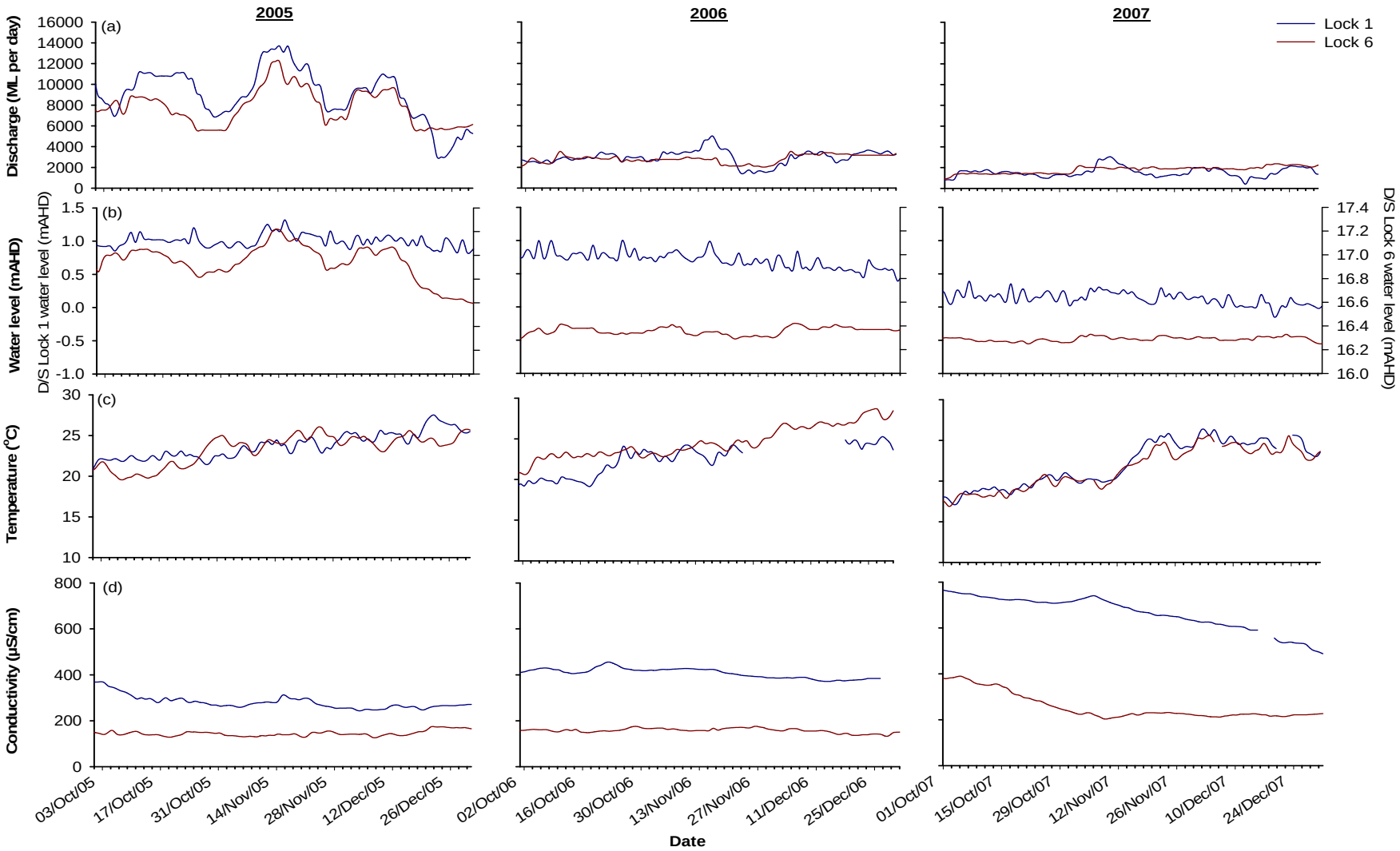

Figure 3. Comparison of the environmental variables between 2005, 2006 and 2007, downstream of Locks 1 and 6, where a) discharge (ML per day), b) water level (mAHD, AHD = Level relative to Australian Height Datum), c) mean daily water temperature (°C) and d) mean daily electrical conductivity (µS cm^{-1} at 25 °C).

4.2 Larval fish assemblage structure

4.2.1 Catch Summary

Catch rates downstream of Lock 1 were greater in all years than those downstream of Lock 6 (Table 1). During 2005, a total of 9,032 fish larvae were collected, of this total, catch downstream of Lock 1 comprised 71.12 %, and downstream of Lock 6 accounted for 28.88 %. During 2006, 35,312 fish larvae were collected, with 69.33 % and 30.67 % collected downstream of Locks 1 and 6, respectively (Table 1). In 2007, a total of 56,228 larvae were collected, downstream of Lock 1, 55.62 % of larvae were collected, compared with 44.38 % downstream of Lock 6 (Table 1).

Eleven species were collected throughout the study, comprising nine native and two exotic. The small-bodied native species, Australian smelt, carp gudgeons, flathead gudgeon, and bony herring, were the most abundant of the larval catch (Table 1, Figure 4). Additional species sampled as larvae included freshwater catfish, Murray cod, golden perch, silver perch and hardyheads (Table 1 and Figure 5). The exotic species were common carp and redfin perch (Figure 6). Species richness was higher in 2005 than 2006 or 2007, due to the presence of golden perch and silver perch larvae (Table 1, Figure 5). Spatially, species richness was higher downstream of Lock 6, species that were absent in the samples from Lock 1 included silver perch larvae in 2005 and freshwater catfish and Murray cod larvae in 2006 and 2007 (Table 1, Figure 5).

Table 1. Total number of fish larvae collected downstream of Locks 1 and 6 during 2005, 2006 and 2007.

Species		D/S Lock 1			D/S Lock 6			Totals		
Common name	**Scientific name**	**2005**	**2006**	**2007**	**2005**	**2006**	**2007**	**2005**	**2006**	**2007**
Small-bodied native										
Australian smelt	*Retropinna semoni*	2,194	12,124	13,760	476	6,508	14,149	2,670	18,632	27,909
Bony herring	*Nematalosa erebi*	1,431	3,031	133	416	1,508	4,933	1,847	4,539	5,066
Carp gudgeon	*Hypseleotris* sp	2,401	3,927	7,771	882	1,185	2,739	3,283	5,112	10,510
Flathead gudgeon	*Philypnodon grandiceps*	373	5,381	9,581	694	1,470	3,106	1,067	6,851	12,687
Hardyhead	*Craterocephalus* sp	9	9	14	2	81	15	11	90	29
Large-bodied native										
Murray cod	*Maccullochella peelii peelii*	5	0	0	6	5	3	11	5	3
Freshwater catfish	*Tandanus tandanus*	4	0	0	30	5	1	34	5	1
Golden perch	*Macquaria ambigua*	2	0	0	58	0	0	60	0	0
Silver perch	*Bidyanus bidyanus*	0	0	0	14	0	0	14	0	0
Exotic species										
Common carp	*Cyprinus carpio*	5	7	9	27	8	4	32	15	13
Redfin	*Perca fluviatilis*	0	3	8	3	60	2	3	63	10
	Total	**6,424**	**24,482**	**31,276**	**2,608**	**10,830**	**24,952**	**9,032**	**35,312**	**56,228**
	Yearly % catch	**71.12**	**69.33**	**55.62**	**28.88**	**30.67**	**44.38**			

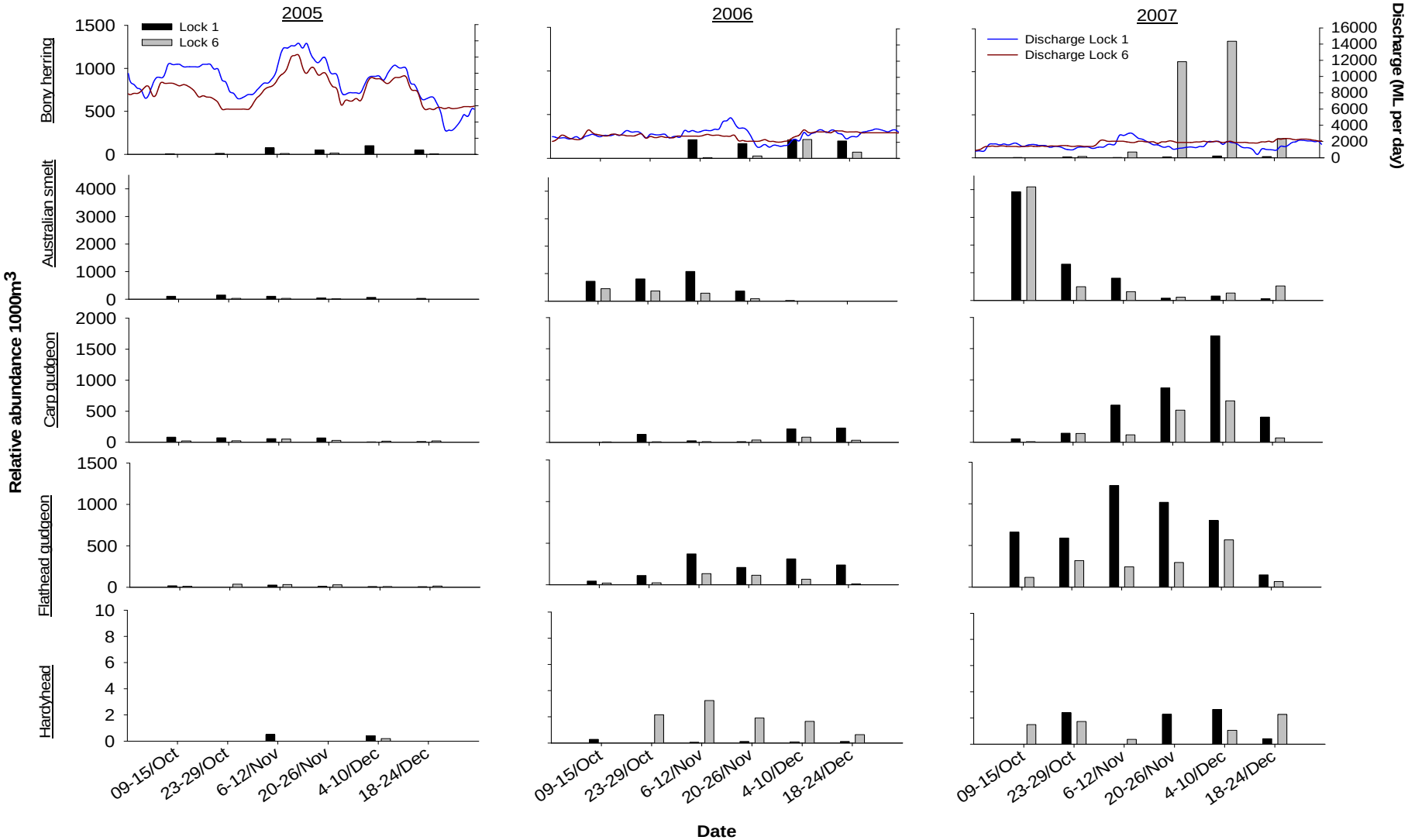

Figure 4. Relative abundance of larvae (fish per 1000 m^3) for the small-bodied native species downstream of Locks 1 and 6 in the River Murray, SA, during 2005, 2006 and 2007. Discharge volume (ML per day), corresponding to the sampling period is plotted for each year in the top graph.

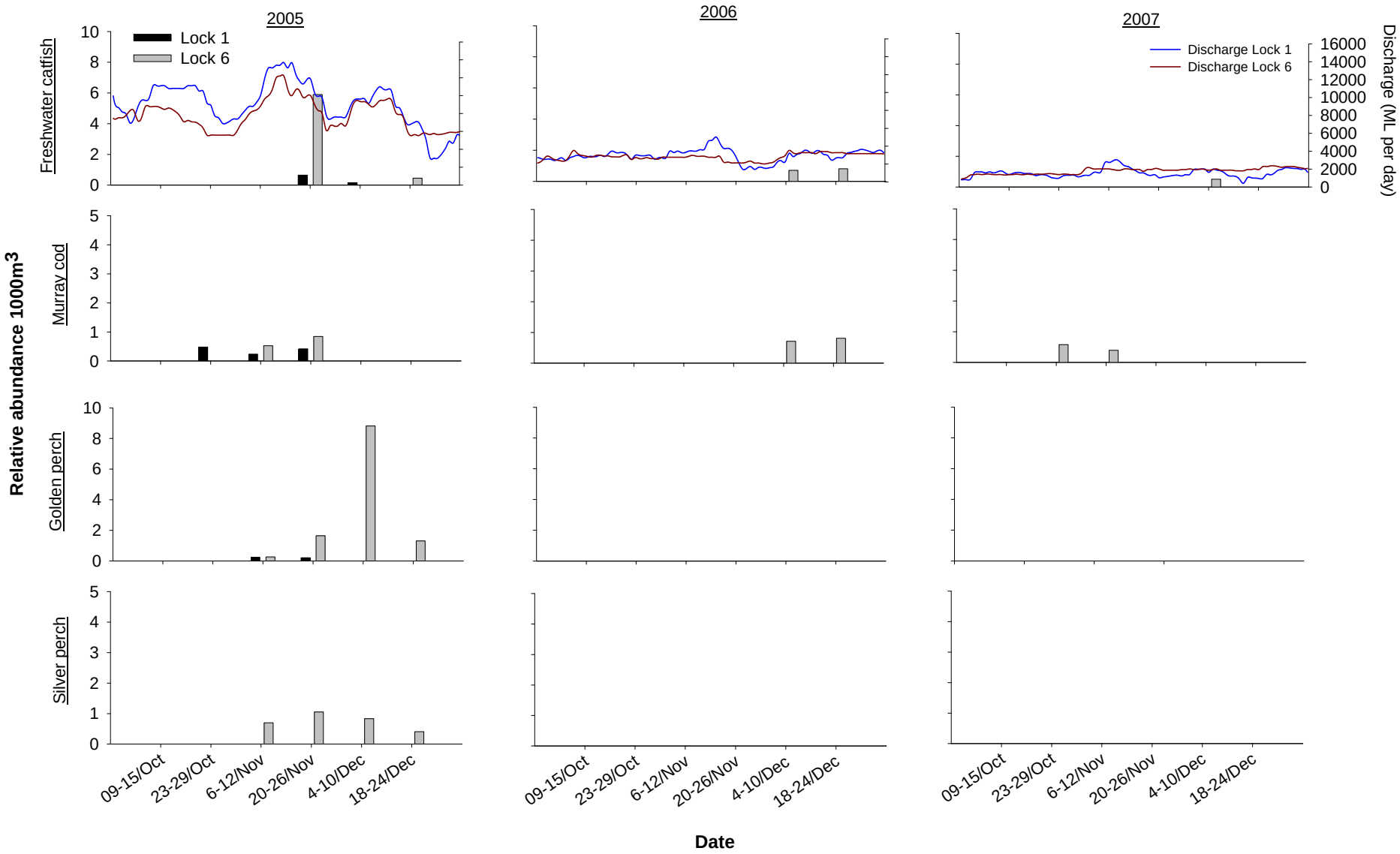

Figure 5. Relative abundance of larvae (fish per 1000 m3) for the large-bodied native species downstream of Locks 1 and 6 in the River Murray, SA, during 2005, 2006 and 2007. Discharge volume (ML per day), corresponding to the sampling period is plotted for each year in the top graph.

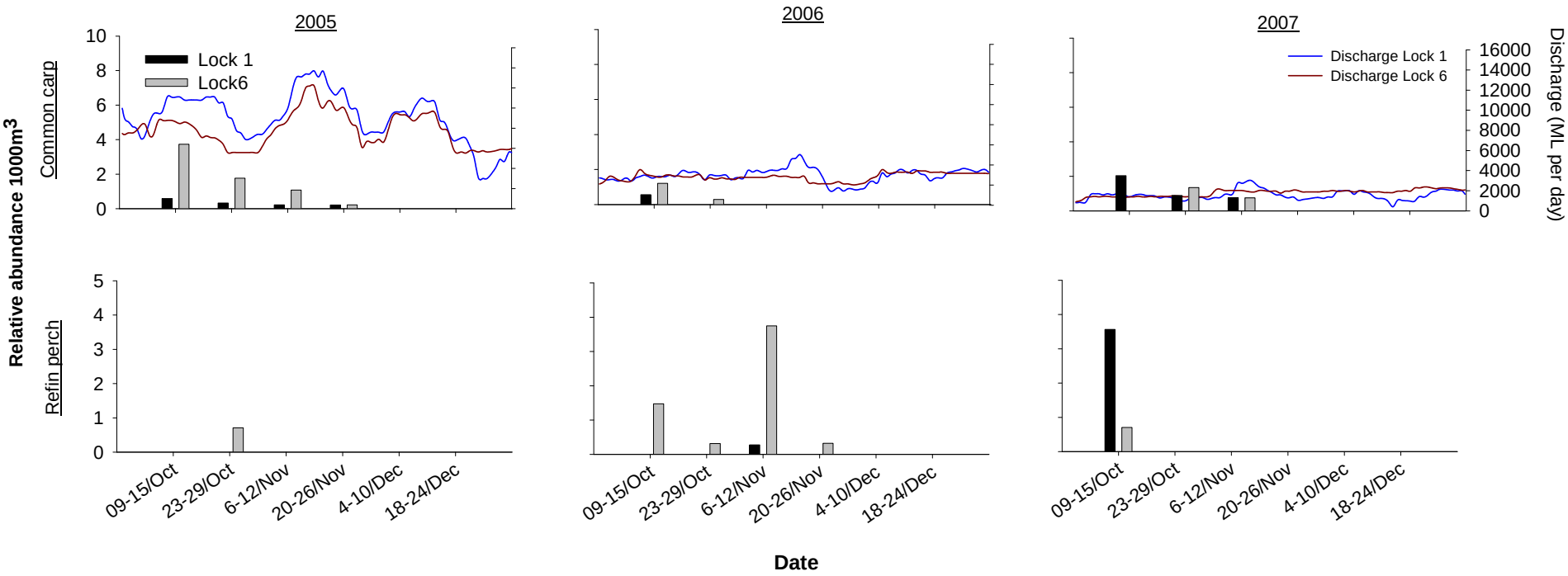

Figure 6. Relative abundance of larvae (fish per 1000 m3) for the exotic species downstream of Locks 1 and 6 in the River Murray, SA, during 2005, 2006 and 2007. Discharge volume (ML per day), corresponding to the sampling period is plotted for each year in the top graph.

4.2.2 Small-bodied native species: inter-annual variation in larval assemblage

PERMANOVA detected a significant year-site interaction (p = 0.001, Table 2) for the small-bodied native assemblage. This indicated that variation in larval assemblage structure between years differed between sites. Pairwise analysis indicated a consistent pattern of significant differences between all years at each site, it was however, the magnitude of these differences that varied, driving the interaction. Given this interaction, further analyses were conducted looking at inter-annual variation at Lock 1 and 6 separately.

Table 2. Two-way multivariate PERMANOVA, for small-bodied native larval assemblages between years and sites. Bold text indicates significant value.

Source of variation	*df*	MS	Pseudo F	*p* (permutation)	Unique permutations
Year	2	9673.5	3.0677	0.086	999
Site	1	3268.7	1.0366	0.444	999
Year x Site	2	3153.3	6.5369	**0.001**	999
Residual	138	482.39			

4.2.2.1 Inter-annual variation downstream of Lock 1

Significant inter-annual differences were detected for the small-bodied native larval assemblage, downstream of Lock 1. The non-metric MDS ordination did not have distinct separations between years, however, 2005 did separating out as a tighter cluster, while 2006 and 2007 were more interspersed and had greater within year variability (Figure 7). The stress value <0.2 indicated that the ordination was a consistent representation of the data (Clarke and Warwick 2001).

SIMPER analysis on the small-bodied native larval assemblage downstream of Lock 1, indicated that Australian smelt, bony herring, flathead gudgeon and carp gudgeon had the most influence on the observed difference in assemblage structure for comparison between all years, accounting for greater than 95% of the differences (Table 3). The difference between years appeared to be driven by an increase in abundance from 2005 to 2007 for identified species, with the exception of bony herring, where abundance was lowest in 2007 (Table 3). The relatively low average dissimilarity indicates that simple differences in species abundances do not fully explain the observed differences between years, more likely driven by a combination of differences in patterns and magnitude of species through time.

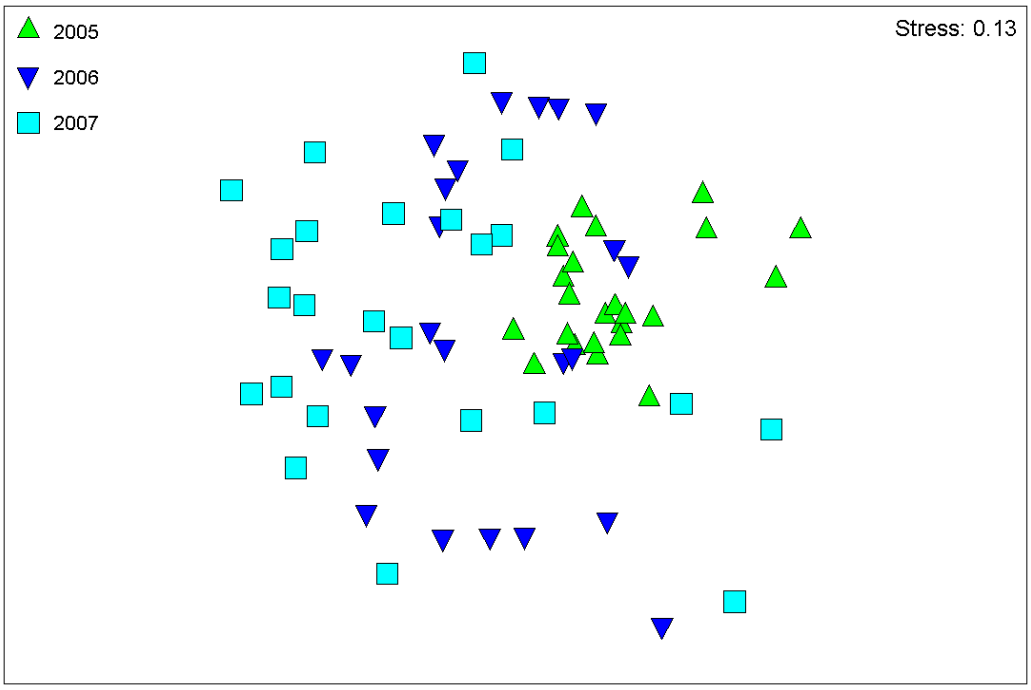

Figure 7. Non-metric MDS ordination (2-dimensional plot) for Lock 1 small-bodied native fish assemblages between 2005, 2006 and 2007 (all trips pooled).

Table 3. SIMPER analysis for the comparison of small-bodied native larval fish assemblages between 2005, 2006 and 2007, downstream of Lock 1. Results are based on standardised fourth root-transformed data. Mean abundance is the standardised value for fish/1000m^3. CR (consistency ratio) indicates species distributions between years, with larger values indicating greater consistency. The contribution (%) indicates the proportion of difference between years (shown by PERMANOVA) attributable to individual species. Mean dissimilarity is expressed as a percentage ranging from 0% (identical) and 100% (totally dissimilar).

Species names	Mean abundance		CR	Contribution (%)	Cumulative contribution
	2005	**2006**		**Mean dissimilarity = 32.68**	
Australian smelt	84.46	527.65	1.52	26.2	26.2
Bony herring	46.5	127.85	1.61	25.94	52.15
Flathead gudgeon	11.49	226.88	1.6	25.55	77.7
Carp gudgeon	48.72	108.71	1.37	18.2	95.89
	2005	**2007**		**Mean dissimilarity = 36.46**	
Flathead gudgeon	11.49	739.43	1.9	32.42	32.42
Australian smelt	84.46	1057.47	1.28	25.9	58.32
Carp gudgeon	48.72	628.92	1.53	23.54	81.85
Bony herring	46.5	10.5	1.22	13.69	95.54
	2006	**2007**		**Mean dissimilarity = 35.67**	
Australian smelt	527.65	1057.47	1.34	26.67	26.67
Carp gudgeon	108.71	628.92	1.35	24.22	50.89
Flathead gudgeon	226.88	739.43	1.43	23.45	74.34
Bony herring	127.85	10.5	1.65	21.01	95.35

Significant correlations between the small-bodied larval assemblage downstream of Lock 1 and the environmental variables were identified (ρ = 0.01). Conductivity (µS/cm) and water level (mAHD) were identified by BIOENV as the strongest contributing environmental variables to observed inter-annual variation in small-bodied native assemblages downstream of Lock 1 (ρ_s = 0.366). Adding additional variables did not increase the correlation factor. Visual representation of these drivers can be seen in Figure 8a-c. Conductivity (µS/cm) increased progressively with each consecutive year, and water level (mAHD) decreased from 2005-07 (Figure 8b, and c, respectively).

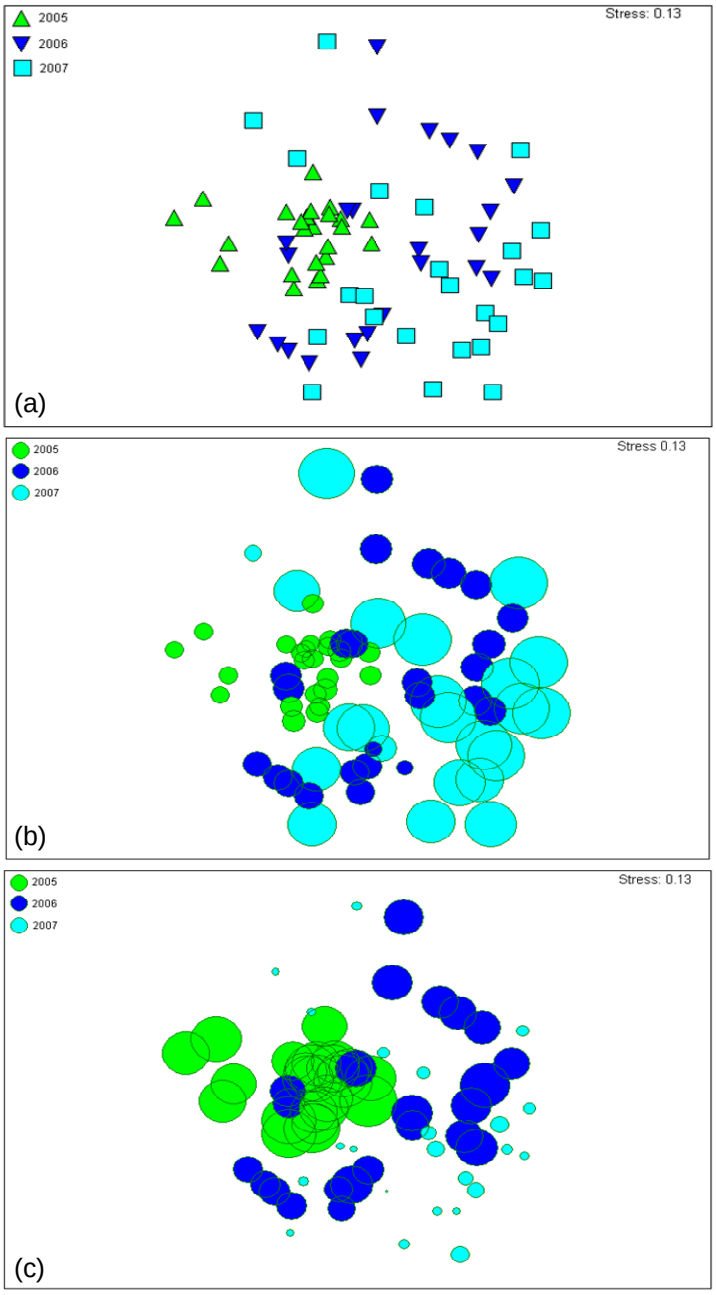

Figure 8. Non-metric MDS ordination (2-dimensional plot) for (a) Lock 1 small-bodied native fish assemblages between 2005, 2006 and 2007 (all trips pooled). Super imposed circles represent the values of (b) conductivity (µS/cm), and (c) water level (mAHD). Larger circles indicate higher values for the raw data.

4.2.2.2 *Inter-annual variation downstream of Lock 6*

Significant inter-annual differences were detected for the small-bodied native larval assemblage downstream of Lock 6. Non-metric MDS ordination, displayed distinct grouping and separation between years (Figure 11).

The difference between years downstream of Lock 6 was again driven by each consecutive year having a higher abundance of each species than the year prior, with the lowest abundances occurring in 2005 (Table 4). SIMPER analysis indicated the inter-annual differences downstream of Lock 6 were consistently driven by abundances of Australian smelt, bony herring, flathead gudgeon and carp gudgeon (Table 4). Hardyheads were only identified as contributing to the difference between 2005 and 2006 (Table 4).

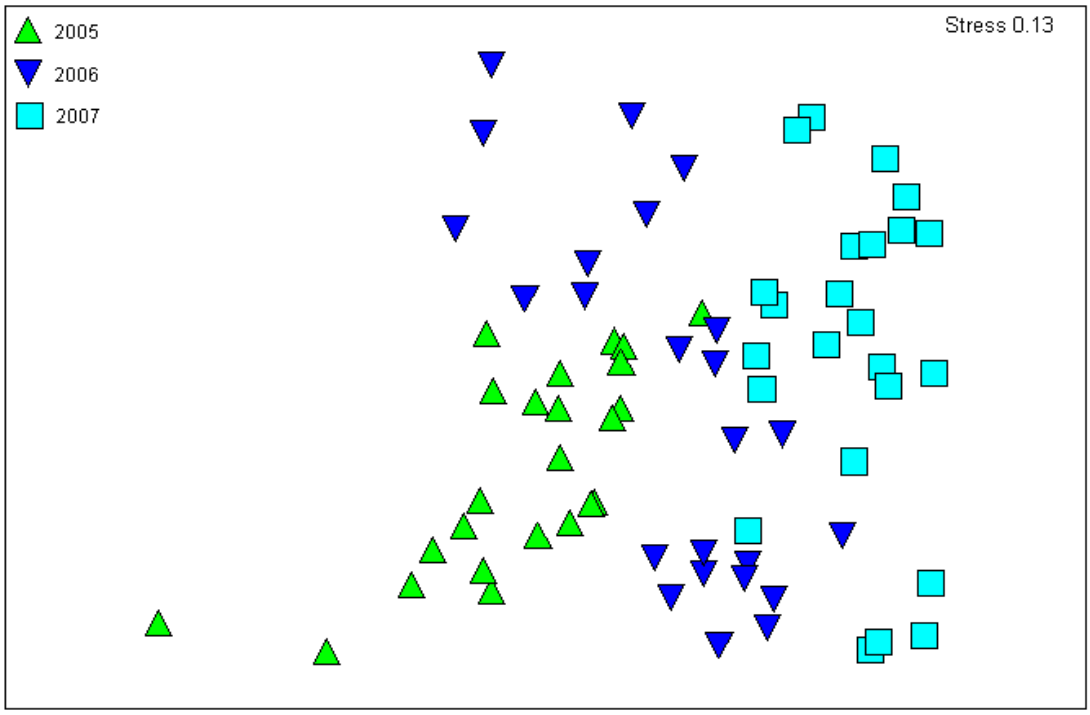

Figure 9. Non-metric MDS ordination (2-dimensional plot) for Lock 6 small-bodied native fish assemblages between 2005, 2006 and 2007 (all trips pooled).

Table 4. SIMPER analysis for the comparison of small-bodied native larval fish assemblages between 2005, 2006 and 2007, downstream of Lock 6. Results are based on standardised fourth root-transformed data. Mean abundance is the standardised value for fish/1000m3. CR (consistency ratio) indicates species distributions between years, with larger values indicating greater consistency. The contribution (%) indicates the proportion of difference between years (shown by PERMANOVA) attributable to individual species. Mean dissimilarity is expressed as a percentage ranging from 0% (identical) and 100% (totally dissimilar).

Species names	Mean abundance		CR	Contribution (%)	Cumulative contribution
	2005	**2006**			**Mean dissimilarity = 33.85**
Australian smelt	16.11	196.39	1.5	33.79	33.79
Bony herring	5.74	51.28	1.17	26.02	59.81
Flathead gudgeon	21.58	64.37	1.17	14.63	74.44
Hardyhead	0.03	1.73	1.38	14	88.45
	2005	**2007**			**Mean dissimilarity = 42.79**
Australian smelt	16.11	970.66	1.45	33.33	33.33
Bony herring	5.74	463.31	1.43	28.45	61.78
Flathead gudgeon	21.58	267.9	1.61	18.09	79.87
Carp gudgeon	26.96	252.35	1.55	16.02	95.89
	2006	**2007**			**Mean dissimilarity = 35.38**
Bony herring	51.28	463.31	1.45	28.7	28.7
Australian smelt	196.39	970.66	1.13	28.27	56.98
Carp gudgeon	30.53	252.35	1.59	17.42	74.4
Flathead gudgeon	64.37	267.9	1.49	16.69	91.09

Significant correlations between the small-bodied larval assemblage downstream of Lock 6 and the environmental variables were identified ($p = 0.01$), however, results should be interpreted cautiously due to the low correlation coefficients. BIOENV identified water level (mAHD) and water temperature (°C) as the strongest correlating environmental variables driving the observed inter-annual variation in small-bodied larval assemblage structure downstream of Lock 6 ($\rho s = 0.300$). Visual representation of these drivers paired with the MDS plot can be seen in Figure 10a-c. Water level (mAHD) decreased minimally with each consecutive year, while temperature (°C) appears relatively consistent between years (Figure 10b and c, respectively).

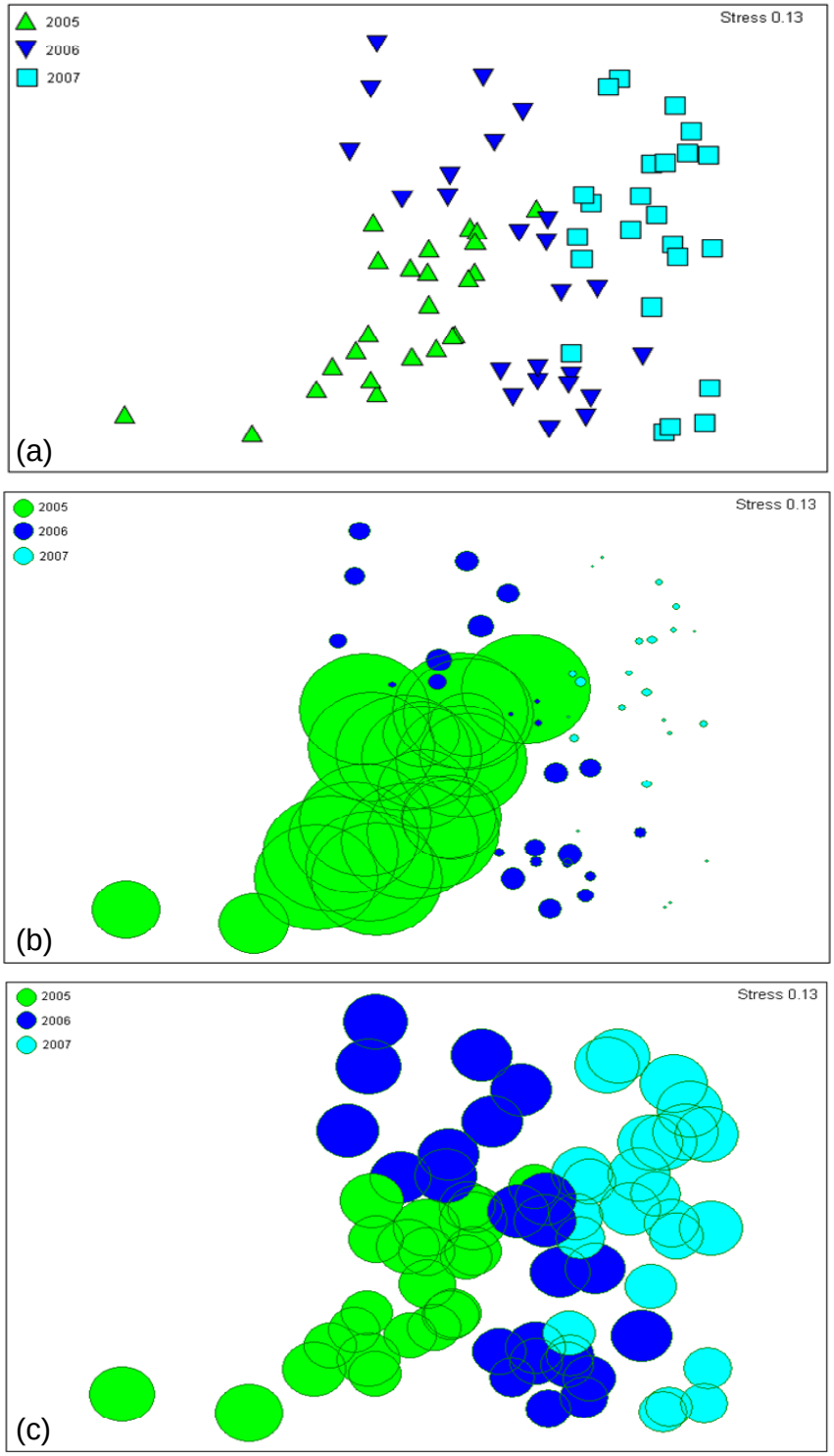

Figure 10. Non-metric MDS ordination (2-dimensional plot) for (a) Lock 6 small-bodied native fish assemblages between 2005, 2006 and 2007 (all trips pooled). Super imposed circles represent the values of (b) water level (mAHD), and (c) water temperature (°C). Larger circles indicate higher values for the raw data.

4.2.3 Large-bodied native species: inter-annual variation

PERMANOVA indicated a significant year-site interaction (p = 0.003, Table 5), thus indicating that variation in large-bodied native assemblage structure between different years differed between sites. Pairwise analysis indicated that at both Lock 1 and 6, the pattern was consistent, with 2005 being significantly different from 2006 and 2007, but no difference between 2006 and 2007, but that the magnitude of differences varied between sites. Given this interaction further analyses were restricted to inter-annual variation within sites.

Table 5. Two-way multivariate PERMANOVA, for large-bodied native larval assemblages between years and sites. Bold text indicates significant value.

Source of variation	df	MS	Pseudo F	p (permutation)	Unique permutations
Year	2	2997.5	1.5786	0.281	998
Site	1	3729.6	1.9642	0.195	998
Year x Site	2	1898.8	3.6861	**0.003**	997
Residual	138	515.11			

4.2.3.1 Inter-annual variation downstream of Lock 1

The non-metric MDS ordination displayed distinct separation for the majority of the 2005 samples from the 2006 and 2007 samples (Figure 11). Whilst some of the 2005 samples group with the 2006 and 2007 samples, this is likely driven by the complete absence of all large-bodied native species in these samples. It should be noted that many of the samples group on top of each other due to the high number of samples with no large bodied species collected. The stress value <0.05 indicates an excellent representation with no prospect of misleading interpretation (Clarke and Warwick 2001).

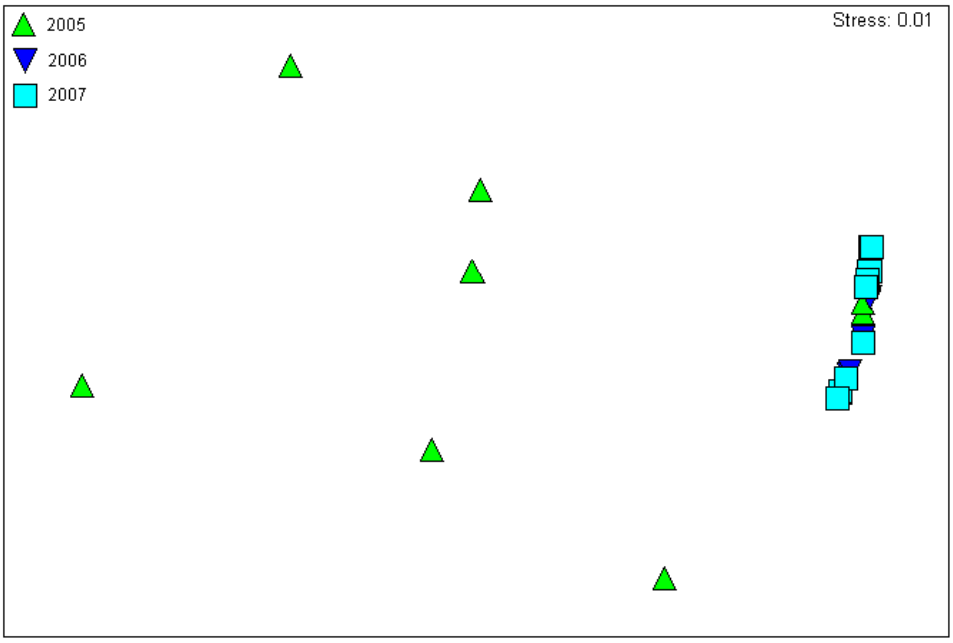

Figure 11. Non-metric MDS ordination (2-dimensional plot) for Lock 1 large-bodied native species fish assemblages between 2005, 2006 and 2007 (all trips pooled). Note sample numbers for all years are equal, however, many points sit directly on top of other points.

SIMPER analysis indicated that differences between 2005/2006, and 2005/2007 were driven by the presence of the large-bodied native species in 2005, which were absent in 2006, and 2007 (Table 6). Murray cod had the highest abundances in 2005 and therefore was the highest contributing species, accounting for 50% of the differences.

Table 6. SIMPER analysis for the comparison of large-bodied native species fish assemblages for differences between 2005/ 2006 and 2005/2007 downstream of Lock 1. Results are based on standardised fourth root-transformed data. Mean abundance is the standardised value for fish/1000m³. CR (consistency ratio) indicates species distributions between years, with larger values indicating greater consistency. The contribution (%) indicates the proportion of difference between years (shown by PERMANOVA) attributable to individual species. Mean dissimilarity is expressed as a percentage ranging from 0% (identical) and 100% (totally dissimilar).

Species names	Mean abundance		CR	Contribution (%)	Cumulative contribution
	2005	**2006**			**Mean dissimilarity = 10.58**
Murray cod	0.2	0	0.5	50.28	50.28
Freshwater catfish	0.15	0	0.37	32.85	83.13
Golden perch	0.09	0	0.29	16.87	100
	2005	**2007**			**Mean dissimilarity = 10.58**
Murray cod	0.2	0	0.5	50.28	50.28
Freshwater catfish	0.15	0	0.37	32.85	83.13
Golden perch	0.09	0	0.29	16.87	100

Significant correlations between differences in the large-bodied larval assemblage downstream of Lock 1 and the environmental variables were identified ($p = 0.01$), however, the correlation values are relatively low, and therefore results should be interpreted cautiously, as there are likely other factors contributing to the observed differences in larval assemblage between years. Discharge (ML/d) was identified as having the strongest correlation ($\rho_s = 0.269$, $p = 0.01$) to the observed inter-annual differences in the large-bodied native fish assemblage downstream of Lock 1. Due to substantial overlapping of samples, these correlations were not displayed visually in an MDS as it was not informative.

4.2.3.2 Inter-annual variation downstream of Lock 6

The non-metric MDS ordination displayed distinct separation between the 2005 samples from the 2006 and 2007 samples (Figure 12). The 2006 and 2007 samples are grouped closely together, as a result of the absence of golden perch and silver perch, with many sites that have no large-bodied native species being displayed directly on top of each other (Figure 12).

The presence of golden perch in 2005 was identified by SIMPER as most strongly contributing to observed differences between 2005/2006 and 2005/2007 downstream of Lock 6 (Table 7). A higher abundance of freshwater catfish in 2005 than in both 2006 and 2007 was the greatest most contribution to the differences between years (Table 7).

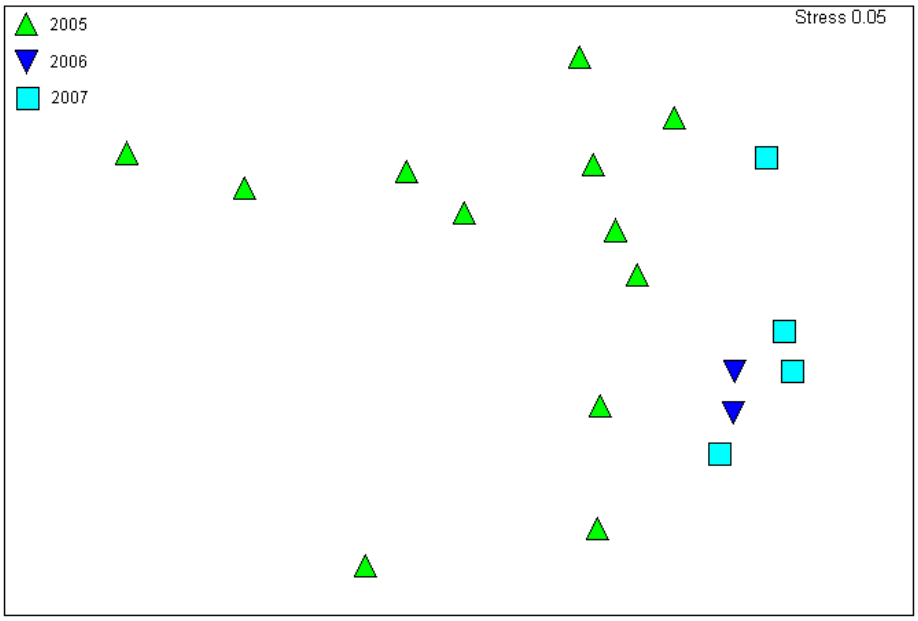

Figure 12. Non-metric MDS ordination (2-dimensional plot) for Lock 6 large-bodied native species fish assemblages between 2005, 2006 and 2007 (all trips pooled). Note sample numbers for all years are equal, however, many points sit directly on top of other points.

Table 7. SIMPER analysis for the comparison of large-bodied native fish assemblages between 2005/ 2006 and 2005/2007, downstream of Lock 6. Results are based on standardised fourth root-transformed data. Mean abundance is the standardised value for fish/1000m^3. CR (consistency ratio) indicates species distributions between years, with larger values indicating greater consistency. The contribution (%) indicates the proportion of difference between years (shown by PERMANOVA) attributable to individual species. Mean dissimilarity is expressed as a percentage ranging from 0% (identical) and 100% (totally dissimilar).

Species names	Mean abundance		CR	Contribution (%)	Cumulative contribution
	2005	**2006**		**Mean dissimilarity = 38.87**	
Golden perch	2.15	0	0.56	36.85	36.85
Freshwater catfish	1.19	0.26	0.54	26.86	63.7
Silver perch	0.51	0	0.57	20.56	84.26
Murray cod	0.25	0.26	0.51	15.74	100
	2005	**2007**		**Mean dissimilarity = 37.38**	
Golden perch	2.15	0	0.56	38.88	38.88
Freshwater catfish	1.19	0.09	0.43	22.05	60.93
Silver perch	0.51	0	0.58	21.9	82.84
Murray cod	0.25	0.25	0.45	17.16	100

Significant correlations between the large-bodied larval assemblage downstream of Lock 6 and the environmental variables were identified ($p = 0.01$). Again, results should be interpreted cautiously given the low correlation coefficients. Discharge volume (ML/d) showed the strongest individual correlation variable ($\rho_s = 0.263$) identified by BIOENV for large-bodied native species downstream of Lock 6. Due to substantial overlapping of samples, these correlations were not displayed visually in an MDS as it was not informative.

4.2.4 Spatial variation in 2007

Cheshire and Ye (*In preparation*) examine the spatial differences in larval fish assemblages between sites downstream of Lock 1 and Lock 6 in detail. These results showed significant spatial differences in the small bodied fish assemblages in both 2005 and 2006. No spatial variation was detected for the large bodied larval assemblage. Given these differences, further analyses were conducted only for spatial variation in 2007 in this report.

4.2.4.1 Small-bodied native species, 2007

PERMANOVA indicated a significant site-trip interaction ($p = 0.048$, Table 8) during 2007. This indicated that variation in larval assemblage between sites differed between sampling trips. Specifically, larval assemblages at Lock 1 and 6 were significantly different for early November (trip 4, $p = 0.034$), late November (trip 5, $p = 0.026$) and early December (trip 6, $p = 0.03$), but not earlier in the year. Further analyses were only conducted on these three trips.

Table 8. Two-way multivariate PERMANOVA, for the 2007 small-bodied native larval assemblages between sites and trips. Bold text indicates significant value.

Source of variation	df	MS	Pseudo F	p (permutation)	Unique permutations
Site	1	1633.5	3.1323	0.055	999
Trip	5	2573.7	4.9352	0.005	998
Site x Trip	5	521.5	1.8001	**0.048**	998
Residual	36	289.71			

The non-metric MDS ordination displayed distinct separation between Lock 1 and 6 for trips 4 and 5 during 2007 (Figure 13a & b). The separation is not as distinct for trip 6, with some of the Lock 1 samples grouping with the Lock 6 samples (Figure 13c).

Flathead gudgeon, bony herring, Australian smelt and carp gudgeons were identified using SIMPER as having the highest contribution to the observed spatial differences between sites, for early November (Table 9). During early November, abundances of flathead gudgeon, Australian smelt and carp gudgeons were higher downstream of Lock 1, and bony herring was higher downstream of Lock 6 (Table 9). The spatial differences identified for late November were driven by higher abundances of bony herring and Australian smelt downstream of Lock 6, and carp gudgeons and flathead gudgeons downstream of Lock 1 (Table 9). A similar pattern was detected for the spatial differences in early December, with higher abundances of bony herring and Australian smelt downstream of Lock 6 (Table 9).

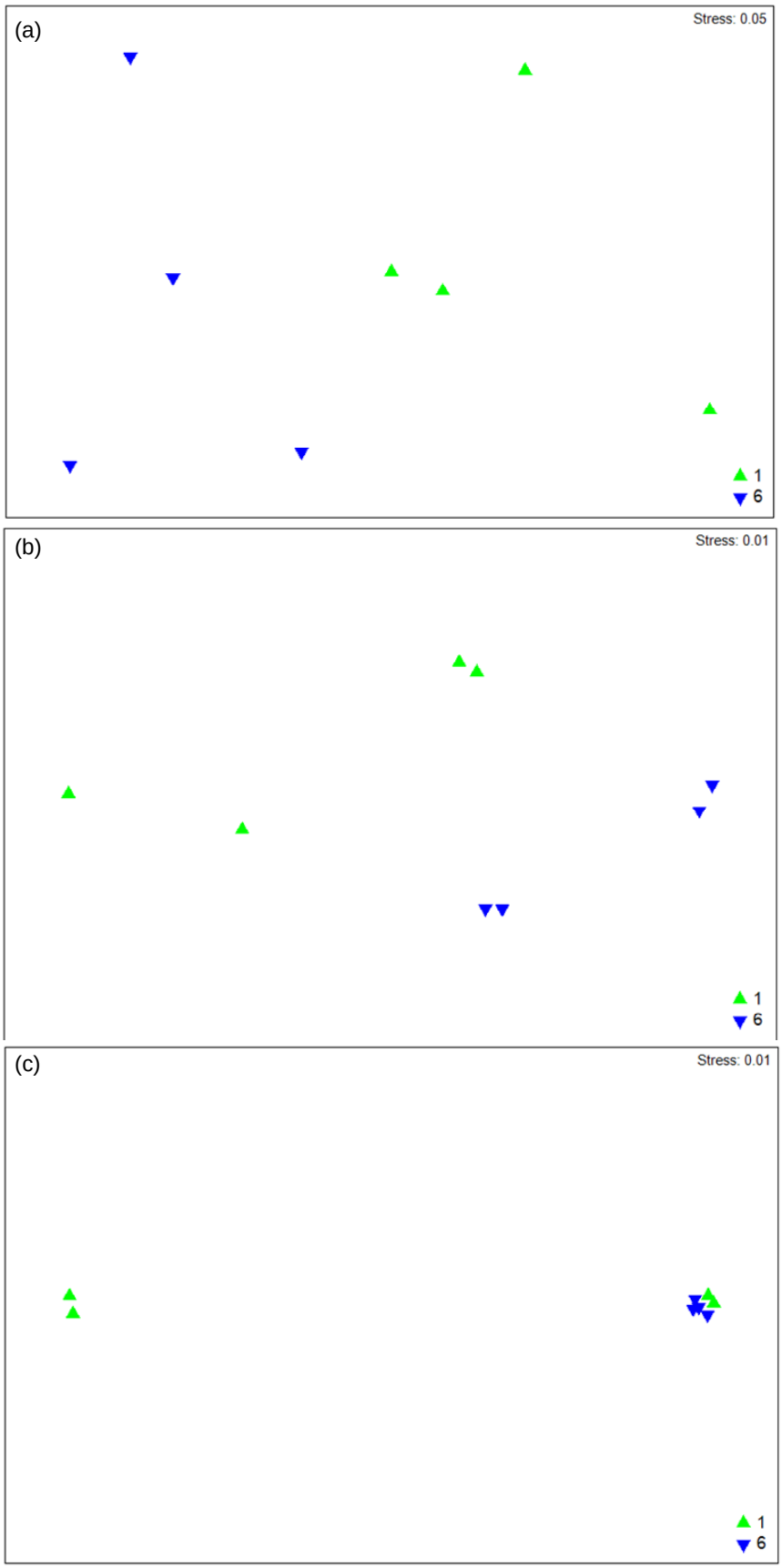

Figure 13. Non-metric MDS ordination (2-dimensional plot) for 2007 small-bodied native species fish assemblages between Locks 1 and 6 for (a) early November (trip 4), (b) late November (trip 5) and (c) early December (trip 6).

Table 9. SIMPER analysis for the comparison of small-bodied native larval fish assemblages between downstream of Locks 1 and 6, for early November, late November and early December 2007. Results are based on standardised fourth root-transformed data. Mean abundance is the standardised value for fish/1000m3. CR (consistency ratio) indicates species distributions between years, with larger values indicating greater consistency. The contribution (%) indicates the proportion of difference between years (shown by PERMANOVA) attributable to individual species. Mean dissimilarity is expressed as a percentage ranging from 0% (identical) and 100% (totally dissimilar).

Species names	Mean abundance		CR	Contribution (%)	Cumulative contribution
Early November	**Lock 1**	**Lock 6**		**Mean dissimilarity = 20.80**	
Flathead gudgeon	1222.01	242.67	2.79	30.79	30.79
Bony herring	3.88	66.95	1.4	24.52	55.3
Australian smelt	806.07	319.93	1.89	20.3	75.61
Carp gudgeon	595.25	118.2	1.08	19.78	95.39
Late November	**Lock 1**	**Lock 6**		**Mean dissimilarity = 27.23**	
Bony herring	9.47	1117.62	3.24	48.64	48.64
Flathead gudgeon	1018.9	294.8	1.69	23.43	72.08
Carp gudgeon	871.5	514.99	1.24	10.8	82.88
Australian smelt	85.03	120.05	1.53	9.66	92.55
Early December	**Lock 1**	**Lock 6**		**Mean dissimilarity = 30.39**	
Bony herring	21.03	1354.83	3.46	37.72	37.72
Australian smelt	159.46	266.42	1.45	23.38	61.1
Carp gudgeon	1702.95	663.34	2.8	17.05	78.16
Flathead gudgeon	799.36	568.15	2.01	15.28	93.43

4.2.4.2 Large-bodied native species, 2007

There was no significant spatial or temporal variation detected for the 2007 large-bodied native species assemblage (Table 10). No further analyses were conducted.

Table 10. Two-way multivariate PERMANOVA, for the 2007 large-bodied native larval assemblages among sites and trips. Bold text indicates significant value.

Source of variation	df	MS	Pseudo F	p (permutation)	Unique permutations	p (Monte-Carlo)
Site	1	543.18	4.3444	0.171	17	0.084
Trip	5	125.03	1	0.475	16	0.509
Site x Trip	5	125.03	0.6421	0.777	20	0.698
Res	36	194.72				

5 DISCUSSION

5.1 Small-bodied native species

The spawning strategies of the small-bodied native species (Australian smelt, carp gudgeons, flathead gudgeons, bony herring and hardyheads) include a combination of main channel generalists, wetland specialists and low-flow specialists, with most species falling into more than one of these categories (Humphries *et al.* 1999; CRCFE 2003; King *et al.* 2003). These species can respond well to a variety of hydrological conditions, will spawn and recruit under all discharge conditions (low flows, within channel flow pulses and high flows) and use a variety of habitats (including the main channel, wetlands and anabranches) (Humphries *et al.* 1999; King 2002; CRCFE 2003). The small-bodied native species were present at both sites in all sampling seasons in high abundances. Never-the less, significant differences in abundance were detected between years for small-bodied native larval assemblages at both downstream of Locks 1 and 6. All species, with the exception of bony herring in 2007, were collected in higher abundances during the low flow years, 2006 and 2007. Bony herring decreased significantly downstream of Lock 1 in 2007, it may be that downstream of Lock 1 the main spawning season was later, perhaps in January or February, however, spawning seasons between Lock 1 and Lock 6 tend to align fairly closely (Cheshire and Ye *In preparation*). Bony herring play an extremely important role in the trophic ecology (food web) of the system, providing a major food source for piscivorous birds as well as the large-bodied species, such as Murray cod and golden perch (Allen 1989; Lintermans 2007). While bony herring is generally highly abundant in the Lower River Murray (Allen 1989; Puckridge and Walker 1990), a decline in this species could have negative effects on a range of other species.

The diverse range of spawning strategies accounts for the high abundance of these species throughout the MDB. Humphries *et al* (1999) classified these species as Mode 3a & b, the larvae are all undeveloped at hatch, have limited mobility and small mouth gapes, resulting in a strong reliance on high densities of appropriately sized prey items. The positive response of these species to the low flow conditions during 2006 and 2007 can be explained by the *low flow recruitment hypothesis* (Humphries *et al.* 1999). This hypothesis indicates that during extended periods of low flow during summer, the smaller volumes of water concentrate the prey to an extent where the densities are high enough to support the larvae present for these species (Humphries *et al.* 1999). The larvae take advantage of this through habitat specialisation into still, warm and shallow littoral zones and backwaters, where the concentration of food items are likely to be highest (Humphries *et al.* 1999). However, there are also potential disadvantages for species spawning and recruiting under low flow conditions such as decreased water quality, increased predation and increased competition for resources, which may be reflected in subsequent recruitment into the juvenile/adult populations (Copp 1992; Humphries *et al.* 1999).

Inter-annual variation in small-bodied native species larval assemblages downstream of Lock 1 was correlated to a combination of water level (mAHD), and conductivity (µS/cm). While downstream of Lock 6, water level (mAHD), and water temperature (°C) were correlated to the significant inter-annual variation in small-bodied native assemblages. However, the correlation values were relatively low, and therefore, despite being significant, the results should be interpreted cautiously. More than likely it suggests that the included variables are not the only drivers for the observed differences in assemblage structure between years.

Water levels in 2005 were highest due to a combination of increased discharge and a water level raising trial (for more detailed discussion see, Cheshire and Ye 2008). However, as a result of the current drought water levels have dropped substantially downstream of Lock 1, where levels were lowest in 2007. Water levels downstream of Lock 6 decreased from 2005 to 2006 and 2007, however, pool level downstream of Lock 6 has generally been maintained throughout the drought. While water level has not been directly linked to spawning for the small-bodied native species, there was a positive response identified in the abundance of larvae present during the lower water level years. This most likely occurs due to the adaptations that allow these species to utilise habitats for larval survivorship, as described in the low flow recruitment hypothesis. However, low water levels can have follow-on effects in terms of opening and closing access to habitat types and water quality deterioration, therefore, further research into the survival of these species into the juvenile/adult stages is required to determine if the increase in larvae is resulting in an increase in numbers recruiting into the adult population.

Temperature is important for spawning and larval survivorship of all MDB species, with most requiring a minimum threshold of 16 °C. Although being correlated with larval assemblage structure downstream of both Locks 1 and 6, temperature did not differ markedly between years and remained within the optimal range throughout the study. It may be that small differences occurred in the pattern of temperature change between years and this may have influenced the assemblage structure through timing of spawning within each region.

Conductivity had substantial inter-annual variation downstream of Lock 1. The upper values that were measured in 2006 and 2007 are not likely to have a negative effect on the spawning and recruitment of any of the small-bodied native species (McNeil *et al. In preparation*). Increased salinity did not appear to have a negative effect on the abundances of the species in the small-bodied native assemblage.

5.2 Large-bodied native species

There are two distinct spawning strategies employed by the large-bodied native fish species, defined as Mode 1 and Mode 2 by Humphries *et al.* (1999). Mode 1 spawners (Murray cod and freshwater catfish), initiate spawning through a combination of circa-annual rhythms and temperature, they do not require an increase in water

discharge (Humphries *et al.* 1999; Humphries 2005; Koehn and Harrington 2005; Koehn and Harrington 2006). It is believed, however, that an increase in discharge will benefit larval survivorship and subsequent recruitment, as stock assessments in South Australia have shown strong year classes associated with higher flows and flooding (Ye *et al.* 2000; Ye and Zampatti 2007). Mode 2 spawners (golden perch and silver perch) are flood/flow cued spawners, and require increases in discharge to initiate spawning and promote larval survivorship (Humphries *et al.* 1999; Mallen-Cooper and Stuart 2003; Gilligan and Schiller 2004; King *et al.* 2005; Ye 2005; King *et al.* 2007). Given the necessity of enhanced discharge for all of these species, they are of most concern when considering the impact of the current drought conditions.

The assemblage during 2005 was significantly different from that in 2006 and 2007 for the large-bodied native species downstream of Lock 1. Murray cod, freshwater catfish and golden perch larvae were only collected downstream of Lock 1 in 2005, no larvae of the large-bodied native species were collected 2006 or 2007. Similarly, downstream of Lock 6 there was significant inter-annual variation in large-bodied larval assemblages between 2005 and both 2006 and 2007, but not between 2006 and 2007. In contrast to Lock 1, this was due to the presence of golden perch and silver perch in 2005, with Murray cod and freshwater catfish being present in all years.

The observed differences in the large-bodied larval assemblages downstream of both Locks 1 and 6, were correlated to discharge volume (ML per day). However, again despite these correlations being significant, the correlation values are relatively low and need to be interpreted cautiously, suggesting variables not included in the analysis may also have a strong effect on structuring inter-annual variation.

Discharge throughout the study period was generally low, although the summer of 2005 was above entitlement allocation. The increased volume in 2005, however, was still well below that required to flood over the river banks (~50,000 ML/d). Thus, the flow event during 2005 was categorised as a within channel flow pulse (Puckridge *et al.* 1998). Change in increased discharge is not believed to induce spawning in wild populations of freshwater catfish (Lake 1967a) or Murray cod (Humphries 2005; King *et al.* 2003; Koehn and Harrington 2005), as they have been shown to spawn in all years independent of flow. Never-the-less, discharge may affect survivorship. In contrast, the absence of golden perch in 2006 and 2007 was most likely a result of the low discharge. These species have been documented to spawn under within channel flow pulses and over bank floods, and withhold spawning if these conditions are not met (Lake 1967; Mackay 1973; Reynolds 1983; Rowland 1983; Harris and Gehrke 1994; Mallen-Cooper and Stuart 2003).

Murray cod and freshwater catfish were not collected in the samples downstream of Lock 1 in 2006 or 2007, however, they were collected downstream of Lock 6 using the same sampling methods and effort in all years. This was not identified as a significant spatial variation due to the low numbers of these species collected

downstream of Lock 6. It should be noted that the absence of collection does not indicate an absence of spawning, it may be that these species were present in very low numbers, which were not detected by our sampling method. Alternatively, being drifting species, Murray cod larvae have been documented to disperse from nests by drifting in flowing waters, the lack of discharge may mean that the flow is not sufficient to adequately disperse larvae, making collection difficult. Regardless, the abundances for larvae of both Murray cod and freshwater catfish were low in all years, at all sites, which is of major concern. Further research and monitoring is required to continue to assess these species and recruitment studies may help to highlight any changes in abundance over the years.

6 CONCLUSIONS

Significant inter-annual and spatial differences occurred in both the small-bodied native and large-bodied native assemblages. There were two distinct spawning responses exhibited by the native fish to differing discharge throughout the study. Small-bodied native species, with the exception of bony herring, were able to respond positively to the low discharge years and increased in abundance during 2006 and 2007. In contrast, the study found large-bodied native species experienced a decrease in species richness, abundance and distribution during the low discharge years. Results suggest large-bodied native species are at significant risk during years of drought and low flow, either from lack of spawning cues, in the case of golden perch and silver perch, or through decreased larval survivorship for Murray cod and freshwater catfish.

Downstream of Lock 1 appeared to be the most affected by drought conditions, with conductivity increasing and water level dropping well below normal pool level. This, however, led to an increase in the abundance of larvae of small-bodied native species. In 2007, bony herring larval abundance decreased downstream of Lock 1, but not downstream of Lock 6. This species is a highly important in trophic ecology, providing food for species at the higher level of the food web; and the decrease could have significant effects on piscivorous bird and other large-bodied native fish species. However, the observed decline in 2007 could be due to inter-annual variation in spawning effort. Further monitoring of the larval abundances is required, and recruitment and population studies could further quantify whether this is of concern. Furthermore, larvae of large-bodied native species were collected in very low numbers throughout the study and no large-bodied native larvae were collected downstream of Lock 1 in 2006 and 2007. This should be of major concern, as these species were collected in this region in 2005, albeit in low abundances, and continued to be collected downstream of Lock 6 in 2006 and 2007.

This study has also demonstrated that a within channel flow pulse may be sufficient to provide suitable conditions for spawning and enhanced survivorship of larval for the large-bodied native species. The large-bodied native species are long lived species, and therefore populations are likely to be able to withstand unfavourable conditions over the short term. However, if current conditions continue, without effective long-term management we may face significant loss of some of our most valuable species. Water management strategies for native fish management need to be further developed, to ensure that during periods of continued low flow water is available to allow for management of flow pulses, thus, providing contingency plans if the current drought conditions persist.

7 MANAGEMENT RECOMMENDATIONS

This study suggests protracted low flow conditions pose a risk to spawning success and larval survivorship of freshwater catfish, Murray cod, golden perch and silver perch. Given their significant conservation value throughout the MDB, these species should be of greatest concern. Under current low flow conditions, species richness, abundance and distribution of the larvae of large-bodied native species is decreased. Furthermore, a within channel flow pulse, as occurred in 2005, may be sufficient to provide suitable conditions for spawning and enhanced larval survivorship of the large-bodied native species. The small-bodied species are all relatively short-lived (1-3 yrs), while species with short generation times are likely to be most seriously effected by changes to the environment; these species have demonstrated the ability to respond positively to low flow conditions. However, it is imperative to continue monitoring these species to ensure that continued prolonged periods of low flow do not have a detrimental effect as time progresses. Regardless of the ability for these species to tolerate low flows, return to a more natural flow regime is important for all native species.

Providing a more "natural" flow regime in regulated rivers, through environmental water allocations (EWA's), is one of the potential management options for the restoration of native fish (Marchetti and Moyle 2001; Arthington *et al.* 2006). However, there are few applied examples and there have been varied levels of success attributed to manipulations of flows (Nesler *et al.* 1988; Travinchek *et al.* 1995; King *et al.* 1998; Freeman *et al.* 2001). These studies all highlight the need to consider attributes of the flow regime, including timing and duration, within individual river systems in order to manage flows for improved fish populations. King *et al.* (2003) suggested that for successful spawning and recruitment of fish during flood conditions a range of environmental conditions would need to occur. These included the coupling of high flows and temperatures, a predictable flood pulse, slow rate of rise and fall, a duration which spanned months and that the magnitude was sufficient to inundate the floodplain. However, all of these factors require further research before optimal management strategies can be outlined. Despite this, some broad suggestions can be made:

- The magnitude of EWA's needs to be significant before benefits will be achieved. The magnitude of these rises need to take into account what volume is required to inundate the floodplain in the surrounding area, as this will have the greatest benefit. The first environmental flow provision in the Murray-Darling Basin started in 1998, and was an allocation of 100 GL per year for watering the Barmah-Millewa Forest. This initial release resulted in negligible benefits, and as a result of modeling studies, the allocation is now accumulated for a number of years providing for a larger periodic release, which will achieve better environmental outcomes (NRE 1999).
- The timing of EWA's needs to consider the most suitable time for a range of benefits to be achieved. A coupling of high temperatures and increased flow have been suggested as most beneficial, however, throughout most of the MDB these two variables are decoupled. Thus, timing should follow the

historically predictable spring flow peaks. In terms of native fish, the best timing needs to match the spring spawning season, therefore allowing native fish to utilise this increase, either through inducing spawning or enhancing survival of larvae and juveniles (King *et al.* 2003).

- The duration of the release of EWA's should also be considered, if the aim is to inundate the floodplain, suitable periods should be defined for this to occur. Brief inundations or rises in water level are unlikely to be of significant benefit, while extended periods may result in poor water quality. Further research is needed before a definite time period can be identified, although a time frame of several weeks to months has been suggested (King *et al.* 2003).
- Patterns of the flow produced from EWA's also need to be considered- it has been suggested that varying the magnitude of the release in order to provide peaks which closely mirror the historical patterns will most likely provide significant benefits to native fish spawning and larval and juvenile survivorship (King *et al.* 2003).
- A slow rate of rise and fall in flow is believed to be the best method, this allows for fish to respond to increases in food and habitat, while a fast rate often has little benefit for aquatic biota (Junk *et al.* 1989; Bayley 1991).
- Frequency of release of EWA's also needs to be considered. The frequency of flooding has been shown to influence the capacity of a population to respond, and this capacity changes depending on the generation times for individual species (Walker *et al.* 1995; King *et al.* 2003). Thus the longevity of target species needs to be taken into account when developing management options.

8 RECOMMENDATIONS FOR FUTURE RESEARCH/MONITORING

- Continued monitoring of the larval assemblages during drought conditions and the lowering of water level downstream of Lock 1 is required, as this will allow for an increased understanding of the potential impacts.
- Continued monitoring downstream of Lock 1 is required to determine if the absence of Murray cod and freshwater catfish larvae in 2006 and 2007 is an ongoing problem, or an artifact of sampling.
- Determination of salinity tolerance in relation to prey availability for Murray cod larvae will help to determine if the increased salinity downstream of Lock 1 is having a negative impact on the larval survivorship in this region.
- Further investigations into the age structures for the large-bodied native species will help identify years with strong recruitment. Comparing strong year classes to environmental variables, such as discharge, salinity and temperature should also be considered, in an effort to increase our understanding of where the potential problems lie for larval survivorship.

- The substantial decrease in the abundance of bony herring larvae downstream of Lock 1 requires further monitoring to determine if this is natural inter-annual variation or a continuing problem as a result of changing conditions due to the drought. This will also provide information for intervention management strategies if required.
- The increase in larvae in years with low flows suggests these species benefit from low flow conditions, however, this needs to be confirmed in the adult population. Recruitment studies for the small-bodied native species looking at age structures and identifying years with strong recruitment would yield significant benefit in determining the effect of ongoing low flows on the recruitment of small-bodied native species

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