River Murray Channel Environmental Water Requirements: Hydrodynamic Modelling Results and Conceptual Models

> Goyder Institute Project E.1.9 Part 2 of 2



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Executive Summary

Background

The Murray Darling Basin Plan requires the development of Long Term Watering Plans (LTWPs) that identify priority assets and functions, ecological objectives and targets for these assets and functions, and environmental water requirements (EWRs) to meet these objectives and targets. The LTWP's will be instrumental in supporting the allocation of environmental water to South Australian River Murray environmental assets. This project contributes to development of the first draft of the Long-Term Watering Plan (LTWP) for the South Australian River Murray water resource plan area.

Prior to commencing this project, some work had been completed for the River Murray in South Australia, including the identification of (i) approximately 130 key environmental assets, and (ii) a suite of targets and Environmental Water Requirements (EWRs) for the broader floodplain and the Lower Lakes, Coorong and Murray Mouth asset. Independent reviews considered that there was:

- a lack of riverine targets to account for hydraulic connectivity and explicitly address the requirements of native fish,
- limited representation of ecological complexity for evaluating the ecological outcomes of hydrologic scenarios, and
- little regard for the requirements of the floodplain landscape between the two asset sites.

This project addresses the need for Ecological Targets and EWRs relating to in-channel habitats and ecological processes (functions).

Scope

Key tasks were to:

- Develop conceptual models outlining hydro-ecological relationships for in-channel assets and functions for QSA (flow to South Australia), ranging from entitlement flows (3,000-7,000 ML day⁻¹) to 40,000 ML day⁻¹,
- Develop relevant Ecological Objectives and Ecological Targets, and
- Identify EWRs required to meet these objectives and targets.

Spatial boundaries for this project are the South Australian Border and Wellington, and the upper discharge limit is QSA = $40,000 \text{ ML day}^{-1}$.

Report Structure

Due to the large amount of information generated by this project, the reporting is presented in two parts. The accompanying report *Ecological Objectives and Targets* (Part 1 of 2) includes information on:

- Background,
- Approach,
- Impacts of river regulation,
- Ecological Objectives and Targets,
- Environmental Watering Requirements, and
- Weir pool manipulation.

The present document *Hydrodynamic Modelling Results and Conceptual Models* (Part 2 of 2), provides information on:

- Hydrodynamic modelling results, and
- Conceptual models.

An overview of the content of both documents is provided below.

Hydrodynamic Modelling

MIKE 21 2D numerical hydrodynamic modelling was used to convert flow (discharge) in the river to spatial distribution in water level and velocity. The model is based on a computational grid that covers the Lock 3-4 reach (91 km). This reach was selected because:

- It has the longest period of pre-lock construction water levels (January 1921 to April 1926), indicating pre-development conditions, and
- There is a significant amount of ecological information available.

This does not mean that results for the Lock 3-4 reach are directly transferrable to all reaches. However, visual assessments of data with previous modelling assessments of velocity in other reaches (e.g. Lock 6–downstream of Rufus River) indicate that the discharge-velocity relationships are sufficiently representative to be fit-for-purpose. The outputs are voluminous, and are presented in this present document. The results include conditions under modelled natural (no weirs) conditions, existing conditions and weir pool manipulations.

Conceptual Models

Detailed hydrological–ecological response relationships (conceptual models) based on a review of existing literature, empirical data and expert knowledge were developed for:

- Ecosystem processes,
- Biofilms,
- Phytoplankton (diatom v. cyanobacterial-dominated communities),
- Fish,
- Aquatic and understorey vegetation,
- Long-lived vegetation (floodplain eucalypts, lignum), and
- Temporary wetlands.

Frogs, waterbirds and invertebrates were not considered individually, as they require habitat within wetlands to achieve targets set for them. The focus is on providing the water to generate and maintain supporting habitat, rather than providing water for species that might utilise the habitat. However, the timing and duration of flows are consistent with the breeding/recruitment requirements of frog species known to occur in the region. The conceptual models are provided in this present document. The conceptual models developed were used to develop Environmental Water Requirements (EWR). These describe flow pulses from 10,000 to 40,000 ML day⁻¹, with metrics for average and maximum return interval, timing and duration. The return intervals were informed by comparisons with modelled natural conditions, existing and potential return intervals.

Impacts of river regulation

A summary of the impacts of river regulation, including alteration of flow resulting from upstream diversions and regulation of flow into South Australia, and the influence of the 11 low level on the Murray between Mildura and Blanchetown is provided (see section 3 of the accompanying report *Ecological Objectives and Targets*). The weirs have little impact on discharge (ML day⁻¹) but have a significant impact on water levels, velocity and surface water-groundwater interactions. The hydrologic components that require consideration in planning and delivery of water; flow magnitude, timing, duration, frequency, rate of change, sequence, connectivity, spatial scale, hydraulic complexity and flow components are also outlined.

Ecological Objectives and Targets

From the understanding developed *via* the conceptual models, Ecological Objectives and Ecological Targets were developed (see section 4 of the accompanying report *Ecological Objectives and Targets*). Where practical, the targets are structured within a SMART (Specific, Measurable, Achievable, Realistic, Time-bound) framework. Well-defined Ecological Objectives provide a clear articulation for managers, scientists, stakeholders and the wider community of what planned deliveries of environmental water are intended to achieve. Clearly defined, specific and measurable Ecological Targets are a means of (i) assessing and reporting condition over time, (ii) determining the need for management action, and (iii) assessing the outcomes of applied management actions. Ecological Targets streamline monitoring and reporting needs for both short-term (event-specific) and longer-term outcomes, and will allow for plain-language reporting on progress towards Ecological Objectives. An example of an Ecological Objective related to Riparian Vegetation, and the relevant Ecological Targets is provided in Table E1.

Ecological Objective	Ecological Target		
Throughout the length of the river channel asset (i.e. SA border to Wellington), establish and maintain a diverse native flood- dependent plant community in	In standardised transects spanning the elevation gradient in the target zone [†] , 70% of river red gums have a Tree Condition Index score \geq 10.		
	A sustainable demographic is established to match the modelled profile for a viable river red gum population in existing communities spanning the elevation gradient in the target zone. [†]		
10,000–40,000 ML day ⁻¹ .	Species from the Plant Functional Group 'flood- dependent/responsive' occur in 70% of quadrats spanning the elevation gradient in the target zone ⁺ at least once every 3 years.		

Table E.1. Example of an Ecological Objective and associated Ecological Targets.

⁺The target zone is the area inundated by flows of 10,000-40,000 ML day⁻¹

Environmental Water Requirements

Seven Environmental Water Requirements (EWR's) spanning the flow range 3,000-40,000 ML day⁻¹ were developed (see section 5 of the accompanying report *Ecological Objectives and Targets*), as it is anticipated there are step-points at which changes will occur in hydrology (e.g. discharge, hydraulics, velocity), lateral connectivity (e.g. area inundated) or ecology (e.g. spawning). A summary is shown in Table E2, with information on the magnitude and variation of discharge (MLday⁻¹), duration of flow, frequency, Maximum allowable return interval and timing (e.g. season). Each EWR is expected to contribute towards achieving the Ecological Objectives and Ecological Targets, and are considered hypotheses to be tested by delivering flows in a manner that achieves variability in timing, duration, frequency and rate of change, and by measuring responses in terms of progress towards the Ecological Targets.

 Table E.2. In-channel Environmental Water Requirements for the lower Murray.

EWR #	Median discharge (ML day ⁻¹)	Discharge (ML day ⁻¹)	Duratio n (days)	Preferred timing	Average return frequency (years)	Percentage of years flow is required	Maximum return interval (years)	Percentage of years that discharge and duration are likely under BP2800 scenario
IC1	10,000	7000 - 12,000	60	Sep-Mar	1.05	95	2	90
IC2	15,000	15,000 -20,000	90	Sep-Mar	1.33	75	2	77
IC3	20,000	15,000 - 25,000	90	Sep-Mar	1.8	55	2	67
IC4	25,000	20,000 - 30,000	60	Sep-Mar	1.7	59	2	67
IC5	30,000	25,000 - 35,000	60	Sep-Mar	1.8	55	2	59
IC6	35,000	30,000 - 40,000	60	Sep-Mar	1.8	55	2	46
IC7	40,000	35,000 - 45,000	90	Sep-Mar	2.1	48	3	31

Conceptual models for in-channel Environmental Water Requirements

Conceptual models for each of the seven EWR's outlined in the table above are presented (see section 6 of the accompanying report *Ecological Objectives and Targets*). These include a description of (i) high level objectives for each EWR, (ii) conditions that may be expected to occur during, or as a result of that flow scenario, and (iii) example hydrographs of how environmental water could be delivered to complement existing flow, mimic the modelled natural hydrograph and achieve the EWR.

Weir pool manipulation

The effects of weirs on in-channel hydraulics are demonstrated by modelling described in the accompanying report, *Hydrodynamic Modelling Results and Conceptual Models*. The analysis shows that most impact on velocity occurs in the lowermost third of the weir pool. An assessment of how weir manipulation could contribute to achieving In-Channel Ecological Objectives and Targets was undertaken (see section 7 of the accompanying report *Ecological Objectives and Targets*). Reinstating variability *via* re-operation of weirs has potential to restore, in part, seasonal patterns of water level, lateral connectivity and hydraulic diversity in impounded areas. Weir-pool lowering is a mechanism to increase water velocity at all discharges. Weir-pool raisings have the capacity to increase water levels and areas inundated, and the impact on upstream water levels is greatest at low flows. Weir-pool raising trials in the lower Murray already have demonstrated positive outcomes for understorey vegetation and riparian trees. However, at flow to South Australia <20,000 ML day⁻¹, the reduction in velocity associated with a weir pool raising and associated impacts on in-stream processes needs to be taken into account.

1. Background

1.1. Basin Plan

The Murray-Darling Basin Authority's (MDBA) Basin Plan establishes a new framework for managing the Basin's water resources, including new roles and responsibilities for the MDBA, the Commonwealth Government and the Basin States. It will be put into effect through a number of mechanisms, including an environmental watering plan as a framework for the use of water to achieve key environmental outcomes. Implementation of the Plan is a priority in the South Australian Department of Environment, Water and Natural Resources (DEWNR) Corporate Plan.

1.2. This project

This project contributes to development of the first draft of the Long-Term Watering Plan (LTWP) for the South Australian River Murray water resource plan area. An LTWP is a requirement under Chapter 8 (Environmental Watering Plan) of the Basin Plan, and will be instrumental in supporting the allocation of environmental water to South Australian River Murray environmental assets. The Basin Plan requires LTWPs to identify priority assets and functions, ecological objectives and targets for these assets and functions, and environmental water requirements (EWRs) to meet these objectives and targets.

The first part of this study, described in *Ecological Objectives and Targets*, includes information on:

- Background,
- Approach,
- Impacts of river regulation,
- Ecological Objectives and Targets,
- Environmental Watering Requirements, and
- Weir pool manipulation.

The present report (Part 2 of 2) provides:

- Hydrodynamic modelling results, and
- Conceptual models.

2. Prior hydrodynamic modelling

Some modelling data are available for the Lock 2 weir pool (Gippel, 2011) and the Lock 5–7 weir pool (Water Technology 2010). In addition, field data collected by Kilsby (2008) and Bice et al., (2013) are available.

2.1. Field data

Kilsby (2008) demonstrated that under low-flow conditions (<5000 ML day⁻¹), velocity and hydraulic complexity are higher in the mid-Murray (NSW)—where the distances between weirs substantially exceed the influence of each weir—than in the weir pool-dominated reaches of the

lower Murray. Long periods of low inflows resulting in extended (multi-year) periods of regulated conditions are likely to have had profound impacts on flow-ecology relationships.

Bice et al. (2013) showed that mean cross-transect velocities increased with discharge, and were highest in the upper weir pools for the Lock 1–2 and Lock 5–6 reaches for flows of 13,000-16,000 and 23,000-33,000 ML day⁻¹. This indicates that the weirs significantly reduce hydraulic complexity for flows up to 33,000 ML day⁻¹. The influence may extend to the flow threshold at which the navigation passes are removed (>50,000 ML day⁻¹) and the structural components of the weirs that remain in the channel are 'drowned'. The authors showed that flows in the range 13,000–16,000 ML day⁻¹ provide hydraulic complexity that was not recorded by Kilsby (2008) at low (<4000 ML day⁻¹) flows. Based on the increase in hydraulic complexity observed at flows >23,000 ML day⁻¹, the authors recommended this as a target for environmental flows.

2.2. Modelled data: Lock 2–3 reach

The impact of instream structures on mean channel velocity is demonstrated by Gippel (2011), who used a simple modelling approach relating surface elevation profiles of the weir pools, water elevation, discharge and bathymetry data to calculate mean cross-sectional velocity (m s⁻¹) for discharge by dividing discharge (m³ s⁻¹) by cross-sectional area (m²). A comparison of mean velocity data generated using this approach for downstream of Lock 3 and upstream of Lock 2 is shown in Figure 2.1. The area upstream of Lock 2 has markedly lower velocity than the area downstream of Lock 3. Downstream of Lock 3, the velocity threshold of 0.18 m s⁻¹ is achieved at flows <6000 ML day⁻¹. Upstream of Lock 2, the threshold is not reached until flows reach 11,000 ML day⁻¹. The velocities merge at flows of 47,000 ML day⁻¹, when water levels on each side of the structure equilibrate.

2.3. Modelled data: Lock 5–7 reach

Modelling was undertaken by Water Technology (2010) utilising MIKE FLOOD software to investigate differences between modelled natural and existing conditions for the reach from Lock 5 to downstream of Rufus River to demonstrate the impact of the instream structures on mean channel velocity. Various scenarios were modelled, including flows from 1000–80,000 ML day⁻¹, with Lock 6 at normal pool height, with –1.0 m lowering, –0.5 m lowering and +0.5 m raising. Selected data are included in section 7.6 of the companion report, *Ecological Objectives and Targets*.

Key observations are that the weirs cause a proliferation of comparatively slow-flowing environments and an increase in flows required to establish moderate and moderate–fast velocities. Under modelled natural conditions, the following indicators apply:

- 79% of the reach would have a mean channel velocity of >0.31 m⁻¹ at 7000 ML day⁻¹. Under existing conditions, 20,000 ML day⁻¹ is required to achieve this threshold.
- 97% of the reach would have a mean channel velocity >0.18 m⁻¹ at 5,000 ML day⁻¹.
 Under existing conditions, a flow of 15,000 ML day⁻¹ is required.

This suggests that the instream structures have increased threefold the discharge required to achieve these velocities in the Lock 5-7 reach.



Figure 2.1. The influence of position within a weir pool on mean cross-sectional velocity. Data are modelled mean channel velocity in the Lock 2 weir pool. Data from Gippel (2011).

3. Hydrodynamic modelling

3.1. Approach

A combination of hydrologic modelling and hydrodynamic modelling was used in this project to assess the relationships between discharge, velocity and depth. This provides a baseline or 'hydrologic foundation' from which to compare natural (pre-regulation) and current conditions. Hydrologic modelling undertaken by MDBA for the Basin Plan was used to assess the frequency of occurrence of events over long time periods (114 years) under different conditions. MIKE 21 2D numerical hydrodynamic modelling was used over shorter periods to convert flow (discharge) in the river to an assessment of spatial distribution in water level and velocity. Input data include land surface elevation, surface roughness, initial water level and eddy viscosity for each grid cell. MIKE 21 resolves the flow equations on the cell-based grid to compute water-level variation and flow in response to inflows and to the various forces acting on a body of water. The model used was developed as part of the *Riverine Recovery Program* (DEWNR 2012), and results were validated against observed river height data. For this study, the model for the reach between Lock 2 and Overland Corner was trimmed so that the upper and lower boundaries were at Lock 4 and Lock 3, respectively. The grid resolution for this model was 15×15 m.

3.1.1. Indicator reach

The Lock 3–4 reach was selected as an indicator reach because:

- it has the longest period of pre-lock construction water levels (January 1921–April 1926), indicating pre-development conditions,
- there is substantial ecological information available, and
- it is a comparatively long reach (91 km).

Visual assessment of data from this project with previous assessments of velocity in other reaches (e.g. Lock 5 to downstream of Rufus River) suggests that the discharge-velocity relationships are sufficient to be fit-for-purpose.

For each scenario, the height of the downstream boundary (Lock 3) was set at fixed height (e.g. 9.8 m AHD for normal pool level), with constant inflow at the upstream boundary. Similar scenarios with different flow rates were simulated—beginning with the lowest flow, the initial height in the river was set to a constant level appropriate to the scenario and the model used to simulate the change in height resulting from the flow. The simulation was continued until the level at Lock 4 had reached equilibrium (about 3 days). Model conditions at the end of each scenario were used as initial conditions ('hot start') for the next simulation in the series.

For pre-development scenarios, the downstream height needed to be set at the 'natural' level that would be present under flow conditions that applied. To do this, a rating curve was fitted to flow and level data. Daily water level data for the river before construction of Lock 3 were available for January 1921–April 1926. Monthly average flow was estimated for this period using the MSN model (data from MDBA). A power-curve relationship was fitted to the monthly average flows and monthly average levels (Figure 3.1).

Four conditions were assessed:

- existing conditions,
- pre-development (natural) conditions,
- -0.25 m weir pool lowering, and
- +0.5 m weir pool raising.

Each assessment considered 11 flow magnitudes, namely 3000, 5000, 7000, 10,000, 12,000, 15,000, 20,000, 25,000, 30,000, 35,000 and 40,000 ML day⁻¹. The modelled surface water levels generated via this process are presented in Table 3.1.



Figure 3.1. Flow–water level relationship at lower model boundary (Lock 3) for pre-development scenarios. $Q = c(h+a)^{b}$ where Q = discharge (MI/day), c = 436.2, h = river height (m AHD), a = -2.46, b = 2.51.

Table 3.1. Downstream rive	r heights used for	pre-develo	pment scenarios.
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Scenario flow	River height			
(ML day⁻¹)	(m AHD)			
3000	4.615			
5000	5.103			
7000	5.483			
10,000	5.947			
12,000	6.210			
15,000	6.560			
20,000	7.059			
25,000	7.488			
30,000	7.867			
35,000	8.211			
40,000	8.526			

3.1.2. Velocity

Modelled velocity values represent the depth-averaged flow for each cell in the 2D model grid. To calculate statistics for the relative proportion of different velocity classes, grid cells outside of the main channel were excluded (e.g. Katarapko Creek). The numbers of grid cells in each velocity class were counted and divided by the total number of cells in the area of interest.

For statistics at a sub-reach scale, a middle thread line was created based on the current GIS representation of the river edges (the midpoint between the banks). This line was used as a reference to define the distance along the stream (relative to Lake Alexandrina). The distance upstream was calculated for each grid cell in the main channel and used to isolate data for statistical analysis. For upper, middle and lower pool statistics, the river was divided into equal thirds based on this distance. For river kilometre quartile plots, main channel cells located between kilometre marks were selected for analysis. Velocity classes in previous assessments of hydro-ecological relationships were applied:

- 0-0.05 m s⁻¹ (very slow),
- 0.05-0.1 m s⁻¹ (slow),
- 0.1-0.15 m s⁻¹ (slow-moderate),
- 0.15-0.18 m s⁻¹ (moderate),
- 0.18-0.25 m s⁻¹ (moderate-fast),
- >0.25 m s⁻¹ (fast).

3.2. Modelling results

3.2.1. Lock 3-4 reach under modelled natural and current conditions

A comparison of modelled natural conditions and existing conditions for normal weir pool heights with flows from 3000–40,000 ML day⁻¹ is shown in Figures 3.1 A-K. Under modelled natural conditions, even at the lowest flows assessed (3000 ML day⁻¹, Figure 3.1 A), a high proportion of the Lock 3–4 reach displayed moderate–fast flowing habitat. At all flows, the impact of the weir on velocity in the upper weir pool is relatively small. In contrast, at low flows (3000–7000 ML day⁻¹, Figures 3.1 A-C), the middle and lower reaches are dominated by slow-flowing habitat. It is only when flows exceed 15,000 ML day⁻¹ (Figure 3.1 F) that substantial proportions of the middle and lower reaches provide moderate and fast flowing habitat.

3.2.2. Influence of weir manipulation

A comparison was undertaken for QSA from 3000–40,000 ML day⁻¹ with Lock 3 at:

- normal weir pool height,
- -0.25 m lowering, and
- +0.5 m raising.

Data are shown in Figures 3.2 A-K. At flows \leq 15,000 ML day⁻¹ (Figures 3.2 A-F), a 0.5-m weir raising has a marked impact on the proportion of moderate–fast flowing habitat in the lower reach of the weir pool. The impact is most pronounced at low flows (< 7000 ML day⁻¹). For interpretation of the data in an ecological context, see the companion report, *Ecological Objectives and Targets*.



Figure 3.1 [A]. The influence of Lock 3 on velocity in the Lock 3-4 reach at QSA = 3000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [B]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 5000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [C]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 7000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [D]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 10,000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [E]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 12,000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [F]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 15,000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [G]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 20,000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [H]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 25,000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [I]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 30,000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [J]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 35,000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.1 [K]. The influence of Lock 3 on velocity in the Lock 3–4 reach at QSA = 40,000 ML day⁻¹. The upper plot is modelled natural conditions; the lower plot is existing conditions. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [A]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = 3000 ML day^{-1} . The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [B]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = 5000 ML day⁻¹. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [C]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = 7000 ML day⁻¹. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [D]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = 10,000 ML day⁻¹. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [E]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = 12,000 ML day⁻¹. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [F]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = $15,000 \text{ ML day}^{-1}$. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [G]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = 20,000 ML day⁻¹. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [H]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = 25,000 ML day⁻¹. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [I]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = $30,000 \text{ ML day}^{-1}$. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [J]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = $35,000 \text{ ML day}^{-1}$. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.



Figure 3.2 [K]. The influence of weir manipulation on velocity in the Lock 3–4 reach at QSA = 40,000 ML day⁻¹. The upper plot shows existing conditions; the middle plot is a -0.25 m weir pool lowering and the lower plot is a +0.5 m weir pool raising. The horizontal line within each box is the 50th percentile (median), and the boxes are bounded by the 25th and 75th percentiles. Middle Thread Distance is river kilometres; error bars are minimum and maximum values.
4. Conceptual models

A generic model of the influence of reinstating a flow regime with characteristics of a natural (undeveloped) system is shown in Figure 2.1 of the accompanying report, *Ecological Objectives and Targets*. That model does not include all process and attributes, but identifies (i) key processes required to achieve outcomes at the higher trophic levels, and (ii) the flow-on benefits of outcomes for vegetation in terms of habitat and food resources.

Other models relating hydrological alteration/restoration v. ecological responses have been developed for a range of ecological assets and processes. These were based on a review of existing literature, empirical data and expert knowledge. They include:

- Ecosystem processes,
- Biofilms,
- Harmful and nuisance algal blooms,
- Fish,
- Aquatic and understorey vegetation,
- Lignum,
- Floodplain eucalypts, and
- Temporary wetlands.

Frogs, waterbirds and invertebrates are not considered individually, as they require habitat within wetlands to achieve targets set for them. The focus is on providing the water to generate and maintain supporting habitat, rather than providing water for species that might utilise the habitat.

4.1. Ecosystem processes

4.1.1. Background

The health of the lower Murray was assessed in the MDBA *Sustainable Rivers Audit* (SRA) (Davies et al., 2008, 2010; MDBA, 2012). River 'health' traditionally is assessed *via* structural measurements of community composition, with functional components including rates of processes that often are not considered (Young et al., 2008). This is critical for outcomes at higher trophic levels. For example, the 'quality' of water (its chemical and thermal properties) is important in addition to its quantity or temporal patterns of flow (Arthington et al., 2010). Nutrients are essential for the growth of organisms, and nutrient and light availability are key factors in the structure and health of aquatic ecosystems. If light does not limit plant growth, nutrient limitation is likely to prevail (Udy et al., 2001). Understanding the composition of allochthonous and autochthonous carbon and nutrients, and interactions between those resources, primary producers (e.g. algae, bacteria, fungi) and consumers (e.g. invertebrates, fish, frogs, birds) is fundamental because anthropogenic activities impact (i) movement of resources within and between habitats and (ii) foodweb structure (Douglas et al., 2005).

Natural Organic Matter (NOM) is a significant energy source in aquatic systems (Wetzel, 2001), and originates from terrestrial sources in the catchments of creeks, rivers and lakes (*allochthonous* organic matter) or from aquatic organisms (*autochthonous* organic matter). Phytoplankton, biofilms (periphyton), bacteria and macrophytes are major autochthonous sources. The main sources of allochthonous material are plant material (Ward, 1986; Robertson et al., 1999; McKnight et al., 2003) and, to a lesser extent, soils (Nelson et al., 1996).

The dominant floodplain eucalypts on the lower Murray floodplain, river red gum (*Eucalyptus camaldulensis*) and black box (*E. largiflorens*), generate a large standing biomass of leaf litter (c. 2500 gm⁻² and 600 g m⁻², respectively: Wallace, 2009). Loadings from understorey vegetation in wetland, grassland and lignum areas typically are c. 400–450 g m⁻² (Wallace and Lenon, 2010), and are a large source of allochthonous organic matter (Glazebrook and Robertson, 1999; Francis and Sheldon, 2002).

When floodwater enters a floodplain there is rapid leaching (within hours) of carbon and nutrients from natural organic material (Baldwin, 1999; O'Connell et al., 2000; Francis and Sheldon, 2002) and soils. This is referred to as the 'Birch Effect' (Scholz et al., 2002; Kobayashi et al., 2008; Banach et al., 2009; Wilson et al., 2010). These resources are rapidly incorporated into microbial and algal biomass (Schemel et al., 2004) and cycled to higher trophic level organisms (e.g. birds, fish) *via* multiple pathways, including micro- and macroinvertebrates. This secondary process is referred to as 'trophic upsurge' (Furch and Junk, 1997; Kern and Darwich, 1997; Geraldes and Boavida, 1999; Scharf, 2002; Talbot et al., 2006; Lourantou et al., 2007).

Existing models

There are several conceptual models describing the relative importance allochthonous–autochthonous NOM and lateral–longitudinal connectivity. These include the *river continuum concept* (Vannote et al., 1980), the *serial discontinuity concept* (Ward and Stanford, 1983; Stanford and Ward, 2001), the *flood pulse concept* (Junk et al., 1989), the *riverine productivity model* (Thorp and Delong, 1994) and the *pulse reserve model* (Noy-Meir, 1973; Reynolds et al., 2004). The models emphasise different processes. Variations in the types of resources and spatio-temporal variability within and between systems means that no one model can be universally applicable (Anderson et al., 1996; Robertson et al., 1999).

None of these models was developed expressly for Australian rivers, but a model for lowland rivers in the southern Murray-Darling Basin might predict that the dominant source of Dissolved Organic Carbon (DOC) would shift dynamically between low-, moderate- and high-flow periods. When flows are confined to the channel, most energy is likely to come from instream sources as the exchange of material between the floodplain and the channel is limited. The importance of allochthonous carbon will increase when overbank flooding mobilises the store of allochthonous material, which increases progressively with the duration of intervals between floods.

Fate of allochthonous and autochthonous carbon

Elevated flows and lateral connections are likely to increase the concentrations of carbon and nutrients in the river (Westhorpe and Mitrovic, 2012; Moran et al., 2013), but a large part of the allochthonous NOM may not be assimilated by higher trophic levels. This is because allochthonous material typically has a high carbohydrate content, resulting in high carbon to nitrogen (C:N) and carbon to phosphorus (C:P) ratios. Elser et al. (2000) showed that median allochthonous C:N ratios were 32:1, compared to 9.6:1 for autochthonous material (median C:P ratios 799:1 and 256:1; N:P ratios 27.3:1 and 26.5:1, respectively). In comparison, median C:N ratios for aquatic invertebrate herbivores are 6.0:1 (range 4–10). Thus, autochthonous NOM sources have median C:N ratios 1.6 times higher than their potential consumers. Allochthonous sources have median C:N ratios that exceed more than fivefold that of their potential consumers. The 'poor' quality of allochthonous material through the food web. C:N ratios in benthic algae may increase as a result of nutrient limitation during low flows (Hunt et al., 2012). Deegan and Ganf (2008) showed that semi-emergent macrophytes were a dominant energy source for macroinvertebrate shredders in the Finniss River (SA). This could reflect preferential selection of food with C:N ratios closest to their body tissue.

The flow-on effects of high C:N ratios are likely to depend on the pathways through which resources are assimilated (Brookes et al., 2005; Douglas et al., 2005). Return flows containing high loads of readily available DOC are one of the most important sources of carbon in lowland rivers (Findlay and Sinsabaugh, 1999; Robertson et al., 1999; Hadwen et al., 2009). This represents a large pool of energy for organisms, such that assimilation of soluble carbon and nutrients from leaf litter (and other allochthonous sources) by planktonic heterotrophs, phytoplankton and biofilms (section 6) may represent the main pathway through which allochthonous NOM enters the food web, rather than *via* shredders and other macroinvertebrates. Sherr and Sherr (1988) proposed that the microbial food web is capable of transporting a significant proportion of carbon to zooplankton.

The relative roles of autochthonous and allochthonous carbon in aquatic food webs vary with the timing and extent of connections between river and floodplain (Oliver and Merrick 2006; Westhorpe and Mitrovic 2012). In the Mitchell River (Qld), Hunt et al. (2012) demonstrated that fish and large crustaceans obtained <50% of their carbon from allochthonous sources in the dry season (low lateral connectivity) and >65% in the wet season (high lateral connectivity). Antecedent conditions, particularly time since the last high flow event, influence the response to a flow event (Westhorpe and Mitrovic, 2012).

4.1.2. Influence of regulation

Regulation of the River Murray has increased the proportion of time under low-flow conditions and decreased lateral connectivity between the channel and floodplain (Maheshwari et al., 1995). This limits opportunities for transfer of energy between river and floodplain wetlands, biasing the riverine food web towards autochthonous sources (Hadwen et al., 2009). Robertson et al. (1999) hypothesised that reduced allochthonous inputs have led to an unnatural dominance of algal production in Australian floodplain rivers.

Oliver and Merrick (2006), Oliver and Lorenz (2007) and Gawne et al. (2007) have shown that during inchannel flows the Murray is energy-limited, with net production near zero. A multi-river study including the Logan, Gwydir and Ovens rivers (Hadwen et al., 2009), undertaken during in-channel flows, showed that respiration of the heterotrophic bacterial community and DOC consumption are limited by the quality of available DOC. In contrast, inputs of allochthonous DOC increase during high flows. Examples include the lower Murray (Aldridge et al., 2012a; Oliver and Lorenz, 2013), lower Macintyre (NSW-Qld border) (McGinness and Arthur, 2011) and Lachlan rivers (NSW) (Moran et al., 2013). Westhorpe and Mitrovic (2012) recorded a positive relationship between discharge and DOC, coupled with slightly higher terrestrial carbon signatures, during floods in the Namoi River (NSW). In the Macintyre River (NSW), a 55% reduction in riverfloodplain connectivity led to a decrease of up to 98% in DOC from anabranch channels (Thoms et al., 2005).

In the lower Murray, nitrogen typically is the limiting macro-nutrient, generating conditions that favour heterocystic nitrogen-fixing species of Cyanobacteria (Baker et al., 2000). Low nutrient concentrations during low-flow periods are due to a combination of low inputs from upstream and high retention of nutrients within the channel (Aldridge et al., 2012a). Water from the Darling River typically has higher nutrient loads than water from the mid-Murray, but when flows from the Darling are a significant proportion of flow to SA the impact of a higher nutrient load on autotrophic production may be partly offset by higher turbidity, reducing the photic depth (Baker et al., 2000). Aldridge et al. (2012a) reported very high retention of dissolved (FRP) and total phosphorus (TP) in the lower Murray during low-flow periods, and low retention during overbank flow. These data indicate that the reach between Lock 9 and Lock 1 is a sink for nutrients during low flows and a source of TP, total nitrogen (TN) and DOC during high flows. In contrast, the lower Murray is a sink for nitrate and nitrite (NOx) during both high and low flows.

The *serial discontinuity concept* (Ward and Stanford, 1983; Stanford and Ward, 2001) predicts that riverine systems will revert towards natural conditions as distance downstream increases from the point of regulation (e.g. a dam or weir). In intensively-regulated systems such as the lower Murray, the distances between weirs are not sufficient to allow recovery. Oliver and Lorenz (2010, 2013) reported that Net Ecosystem Production (NEP) typically is zero and planktonic processes dominate in flowing reaches, including locations downstream of weirs. In contrast, NEP is often negative in weir pools, with non-planktonic processes increasing in importance. This difference was attributed to accumulation of organic material, and suggests that weir pools are functioning as productivity 'sinks' by restricting downstream transport of resources. Stabilisation of water levels is also considered to have changed the dominance of macrophytes and composition of biofilms in the lower Murray (section 6).

4.1.3. Influence of management

Relationship between discharge and concentrations of carbon and nutrients

Robertson et al. (1999) predicted that a flood covering 44 km² in one year would provide as much allochthonous carbon as from autochthonous sources (i.e. phytoplankton). Gawne et al. (2007) considered that this effect could be produced by a smaller flood (34 km²). The concentration and quality of carbon mobilised are positively correlated with the inter-flood duration (Baldwin and Mitchell, 2000; Wilson et al., 2011). DOC concentration relative to hydrology is generally considered to follow a 'clockwise hysteresis' where, for a given discharge, concentrations on the rising limb of the hydrograph are higher than on the falling limb (Hope et al., 1994; McGinness and Arthur, 2011). However, this is not always the case. McGinness and Arthur's (2011) data for the lower Macintyre River (NSW-Qld border) show anti-clockwise hysteresis, where DOC concentrations peak on the falling limb due to return flows from ephemeral anabranch channels. The authors proposed that the duration of flow may determine the direction of hysteresis, such that high flows of long duration (>15–20 days) are likely to deliver most carbon on the rising limb, and high flows of short duration are likely to deliver peak concentrations on the falling limb. The potential to mitigate the return of hypoxic blackwater with falling discharge should be considered.

Potential influence of in-channel flows on carbon and nutrient loads

Flows to South Australia (QSA) in the range 35,000–45,000 ML day⁻¹ represent the onset of substantial lateral connectivity with the lower Murray floodplain. Organic loadings are not likely to be as high during inchannel flows (QSA <40,000 ML day⁻¹), as they are for flows that inundate the mid-Murray floodplain, where tree density is relatively high. However, this needs to be interpreted with caution. Oliver and Lorenz (2013) presented data from the 2010–11 flood showing that flows \geq 20,000 ML day⁻¹ can increase DOC and deplete dissolved oxygen (DO). DOC and nutrient data from Lock 5 and Lock 3 indicate episodic increases in carbon and nutrient concentrations during in-channel flows (Figure 4.1).

Assessments at Chowilla Floodplain show that returns of Environmental Water Allocations (EWAs) from managed wetlands to the creek system generate a measurable carbon and nutrient pulse and stimulate heterotrophic activity in the receiving waters (Wallace and Lenon, 2010; Wallace, 2013a). Thus, marked changes in carbon and nutrient concentrations may occur without inundating river red gum or black box woodlands. Further, ephemeral channels that connect at sub-bankfull flows are important sources of NOM in other lowland rivers in the Murray-Darling Basin (McGinness and Arthur, 2011).

Modelling of the extent of inundated areas at flows from 20,000 to 100,000 ML day⁻¹ indicates that the area of black box inundated is not responsive to flows <40,000 ML day⁻¹ (section 4.7). The flow v. inundated area

relationship for river red gum and ephemeral wetlands (section 4.8), shown in Figure 4.2, indicates a stepchange in areas inundated between 25,000–30,000 ML day⁻¹.

An estimate of the load of carbon and nutrients released from inundation of wetlands and low-elevation floodplain areas can be made using mesocosm data from Brooke et al. (2007) (see Table 4.1 for release values). Mean leaf litter loading recorded was 2564 ± 506 g m⁻² in river red gum woodland and 475 ± 224 g m⁻² in ephemeral wetland/flood-runner channels. The resultant loads of Filterable Reactive Phosphorus (FRP), Total Phosphorus (TP) and Dissolved Organic Carbon (DOC) for flows of 20,000–40,000 ML day⁻¹ are shown in Figure 4.3. The step-change in FRP load between flows of 25,000–30,000 ML day⁻¹ is correlated with the aforementioned step-change in the area of ephemeral wetlands inundated.



Figure 4.1. Change in DOC and NO_x and discharge: flow to SA (QSA) and discharge from Lake Victoria (Rufus River).





Change in Total Phosphorus (TP) and Filterable Reactive Phosphorus (FRP) and discharge: flow to SA (QSA) and discharge from Lake Victoria (Rufus River).

	e N	phemeral wetlands		river red gum woodlands			
Resource	mg per g leaf material	mg per m ²	tonnes per ha	mg per g leaf material	mg per m ²	tonnes per ha	
Filterable Reactive Phosphorus (FRP)	0.39	185.3	1.85	0.04	102.6	1.03	
Total Phosphorus (TP)	0.59	280.3	2.80	0.12	307.7	3.08	
Total Nitrogen (TN)	1.74	826.5	8.27	1.61	4128.0	41.28	
Nitrate + Nitrite (NOx)	0.19	90.3	0.90	0.51	1307.6	13.08	
Dissolved Organic Carbon (DOC)	12.7	6032.5	60.33	23.8	61,023.2	610.23	

Table 4.1. Release of carbon and nutrients from leaf litter at Chowilla Floodplain. Mean leaf litter loading was 475 ± 224 g m⁻² in ephemeral wetland/flood-runner channels and 2564 ± 506 g m⁻² in river red gum woodland areas. Data from Brookes et al. (2007).



Figure 4.2. Flow-area inundation plot for river red gum woodland and ephemeral wetlands. Note the step-change in area of wetlands inundated between flows of 25,000–30,000 ML day⁻¹.



Figure 4.3. Potential loads (kilotonnes) of [A] Filterable Reactive Phosphorus (FRP) and Total Phosphorus (TP) and [B] Dissolved Organic Carbon (DOC) related to flow to SA (QSA). Note different scales on Y axes.

Dissolved oxygen

During high flows, allochthonous carbon inputs may stimulate heterotrophic metabolism and create an oxygen demand that exceeds the rate of resupply of oxygen. Events where the surface water contains enough DOC to discolour the water sufficiently to resemble dark 'tea' are 'blackwater' events, and often are associated with low dissolved oxygen (DO) levels (Howitt et al., 2007; Meyer, 1990). Two factors that determine outcomes are water temperature and carbon loading (Baldwin and Wallace, 2009), which depends on inundation extent. Additional factors include flood timing (Baldwin, 1999; Watkins et al., 2010a, b) and whether or not the accumulated plant material litter has been previously flooded (O'Connell et al., 2000). Summer flooding is problematic as (i) peak litter fall for eucalypts is in summer (Briggs and Maher, 1983) and (ii) a 10°C increase in water temperature approximately doubles the rate of oxygen depletion (Howitt et al., 2007). Therefore, the rate of oxygen depletion will vary between flows in late winter and early summer.

Hypoxia and anoxia (low and zero DO levels, respectively) are a concern as tolerances are species- and life-stage specific, and changes in DO can have significant impacts on biodiversity (Ekau et al., 2010). Hypoxia is associated with fish kills (Erskine et al., 2005), disruption of

endocrine systems (Wu et al., 2003) and embryonic development (Shang and Wu, 2004) and survival and hatch rates of fish (Hassell et al., 2008) and degradation of lotic macroinvertebrate communities (Feminella et al., 2003; Walsh, 2002; Walsh et al., 2001) and wetlands (Spieles and Mitsch, 2003). Anoxia may release sediment-bound manganese, iron (Davison, 1993), ammonium (Lawrence & Breen, 1998; Boulton & Brock, 1999; Morin & Morse, 1999) and phosphorus (Mortimer, 1941; Laws, 1993; Martinova, 1993), with conversion of dissolved organic nitrogen to ammonia and nitrate (Harris, 2001) and accumulation of redox-sensitive compounds from sediments (e.g. Baldwin & Mitchell, 2000; Dahm et al., 2003), some of which (e.g. ammonium, sulfide) are toxic to aquatic organisms (Vismann, 1996; Hickey & Martin, 1999).

Open water productivity

Gross Primary Productivity (GPP, total autotrophic conversion of inorganic carbon to organic carbon), Ecosystem Respiration (ER, total oxidation of organic carbon to inorganic carbon by autotrophs and heterotrophs) and Net Ecosystem Production (NEP = GPP – ER) change as the availability of resources changes (Staehr et al., 2012). Over year–decadal time frames, GPP and ER should be in balance, but imbalances may develop over short (days–weeks and seasonal) periods. Variability in inter- and intra-seasonal discharge alters in-channel hydraulic conditions (water column depth, velocity, turbulence, mixing depth), and lateral and longitudinal connections will alter the availability of carbon and nutrients to fuel metabolic pathways. Water clarity (turbidity) and season will influence outcomes *via* photic depth, water temperature, leaf litter loads, day length and sunlight intensity.

Several studies have shown that the Murray (Oliver and Merrick, 2006; Oliver and Lorenz, 2007, 2013) and other lowland Murray-Darling Basin rivers (Bunn et al., 2003; Hadwen et al., 2009; Moran et al., 2013) are energy-constrained (NEP near zero) during periods of low flow, and are then mainly reliant on phytoplankton production. Zero NEP implies that all of the inorganic carbon converted to organic material by autotrophs is respired within the system, and there is no 'gain' in biomass (Staehr et al., 2012). Periods of high discharge and increased allochthonous inputs would result in increased rates of ER (Oliver and Lorenz, 2013) and negative NEP. This would imply a higher rate of utilisation and incorporation of organic carbon into biomass by heterotrophs (Dodds and Cole, 2007).

Oliver and Lorenz (2013) presented data from the 2010–11 flood to show that declining DO during bankfull or overbank flows may be driven partially by return flows that are low in DO due to heterotrophic metabolism and/or chemical oxygen demand on the floodplain. Hence, low DO in the channel may not be a simple reflection of the supply of allochthonous material from the floodplain being metabolised within the channel. In addition, riverine oxygen peaks that were out of sequence with day length (maximum DO during dark conditions) were interpreted as a result of photosynthesis on the floodplain rather than in-channel, with the delayed oxygen signal due to high DO in return flows and the travel time of water between floodplain and monitoring station. This indicates that during high flows large floodplains are a source of (i) allochthonous NOM accumulated and stored during inter-flood periods, and (ii) new autotrophic production. This pattern of shallow zones tending to net autotrophy and subsidising respiration in deeper zones occurs in a variety of aquatic ecosystems (Staehr et al., 2012).

Trying to ascertain the input of allochthonous resources delivered into the channel *via* lateral connectivity using only high temporal resolution DO data is likely to be misleading. As

mentioned, Oliver and Lorenz (2013) attributed a large proportion of changes in in-channel DO to processes on the floodplain. Therefore, changes in open-water metabolism may occur in the absence of an increase in the bulk concentration of DOC. Further evidence is provided by Wallace (2013a). During a release of environmental water from a floodplain wetland to an anabranch creek, there was no substantial increase in DOC but a marked downstream increase in Biochemical Oxygen Demand (BOD₅). The increase in BOD₅ demonstrated that the wetland was a major source of (i) labile organic material that was rapidly assimilated and stimulated instream heterotrophic activity, (ii) phytoplankton, exerting a respiration demand under the dark conditions of the BOD₅ bioassay, or (iii) a mixture of both. An increase in both respiration and GPP measured downstream of the return flows supports the latter suggestion.

Source of water

The source of environmental water may alter outcomes, and information on both the source and the delivery mechanism (i.e. in-channel controlled delivery of water from extended storage *versus* a rain-driven flow peak allowing lateral connectivity with upstream floodplains) is likely to be useful. Upstream storages can be a sink or source of nutrients. The associated changes in speciation of chemicals can alter phytoplankton community structure at downstream sites (Baldwin et al., 2010) with flow-on effects on primary productivity and downstream food webs (Burford et al., 2011). The Darling typically has higher nutrient loads than the Murray (Baker et al., 2000). Aldridge et al. (2012a) demonstrated that during low flows nutrient inputs from Lake Victoria can be higher than from flows from the mid– and upper Murray. Temporary storage of Darling water in Lake Victoria may account for this difference.

Turbidity in the Darling usually is high (>100 NTU) (Sherman et al., 1998; Oliver et al., 2000), and flood waters from the Darling can reduce the photic depth in the lower Murray to <0.2 m (Mackay et al., 1988), limiting the growth of aquatic plants (Brookes et al., 2009). In the late 1980s, water in the lower Murray included disproportionate contributions from the Darling, *via* Lake Victoria, and turbidity was high. When this management practice stopped, turbidity decreased and there were dramatic recoveries in the distribution and diversity of littoral plants and associated invertebrates (Walker et al., 1994; Blanch et al., 1999b, 2000b).

There is a marked difference in the microfauna from the Darling and Murray. The Darling River has a riverine (potamoplankton) microfaunal community dominated by rotifers, which complete their life cycles within days. The Murray community is dominated by lacustrine (limnoplankton) species, including micro-crustaceans (copepods, cladocerans, ostracods) that may take weeks to complete their life cycles (Shiel et al., 1982). Further, the composition of microfauna varies between storages with short (e.g. Lake Mulwala) and long (e.g. Lake Hume) retention times (Brookes et al., 2009). In November 2005–April 2007, a period of sustained drought, the residence time of water in Lake Hume was 150-350 days (Cartwright, 2010).

Zampatti and Leigh (2013a) suggested that a cohort of golden perch spawned during in 2009–10 may have been spawned in the lower Darling following a rise in discharge, and that larval drift and/or downstream movement resulted in the fish colonising the lower Murray. Therefore, a mix of flows from different sources may be required to achieve some ecological outcomes. Reregulating water from high flows and upstream flooding *via* Lake Victoria is not desirable, from an ecological perspective, as the lake is likely to be a sink for resources and propagules that drive outcomes in the lower Murray (Mallen-Cooper et al., 2011).

Return flows of water with high resource loads

As floods recede, a large proportion of the floodplain/wetland production assimilated by consumers is transferred to the receiving water as invertebrates, fish and other animals move with the return flows (Douglas et al., 2005). Returning water that contains a high biomass of organisms and increased nutrient loads is likely to improve fish recruitment in channel habitats (Balcombe et al., 2007; King et al., 2009b; Meredith and Beesley, 2009). Assessments of the microinvertebrate community in Lake Limbra (Chowilla Floodplain) and the adjacent creek system during and following the 2010–11 flood recorded mean densities of 2104 (± 214) individuals per litre, with 49 species represented, compared to 572 (± 55) individuals per litre, with 36 species represented, in the ephemeral lake and permanent creek respectively (D. Furst, unpublished).

4.1.4. Environmental thresholds

The ANZECC Water Quality Guidelines (2000) specify an environmental trigger of 85% saturation for dissolved oxygen in lowland rivers in south-eastern Australia. Section 9.14 (p. 85) of the Basin Plan specifies a target of \geq 50% saturation. Concentrations below 3 mg L⁻¹ are likely to cause stress to fish (Ekau et al., 2010). Although larvae and young-of-year juveniles may survive at DO levels as low as 20% saturation (e.g. 1.8 mg L⁻¹ at 20°C), growth is restricted (Gehrke, 1988, 1991) and fish are likely to die at DO <1.5 mg L⁻¹ (Miranda et al., 2000). A critical threshold of 4 mg L⁻¹ has been identified during environmental watering (Wallace and Lenon, 2010). The influence of water temperature on solubility of oxygen, the 4 mg L⁻¹ threshold and the Basin Plan target (50% saturation) is represented in Figure 4.4.



Figure 4.4. The influence of water temperature on solubility of dissolved oxygen in fresh water. Note that solubility decreases as salinity increases. The Basin Plan target is \geq 50% saturation. The horizontal line is a 4 mg L⁻¹ critical threshold identified by Wallace and Lenon (2010). Data from Eaton et al. (1995).

4.2. Biofilms

4.2.1. Background

Biofilms are microorganisms (algae, bacteria, fungi) that adhere to submerged surfaces including inorganic (e.g. rocks, gravel, sediment) and organic substrates (e.g. wood, aquatic macrophytes). They have an important biogeochemical role, altering nutrient cycles and providing food to invertebrates and other animals (Burns and Walker, 2000a; Vink et al., 2005; Watts et al., 2009b). They have short generation times and are highly responsive to changes in flow over short periods (Watts et al., 2009b). Light and flow regimes are drivers controlling succession. In low-light and low-flow conditions, biofilms typically are dominated by heterotrophs (bacteria and fungi). Within the photic zone, autotrophs (algae) dominate (Lock et al., 1984; Burns and Ryder, 2001).

In systems where there are variable water levels and flows, biofilms are maintained in early- to mid-successional stages, dominated by diatoms and unicellular algae (Steinman and McIntire, 1990). Low variability in water depth and stable photic zones will promote late-successional biofilms, and where light is >1% of the surface irradiance filamentous green algae and Cyanobacteria dominate (Burns and Walker, 2000b). The balance between heterotrophic and photosynthetic organisms is also influenced by nutrient availability and physical disturbances such as scouring, grazing and desiccation (Burns and Ryder, 2001; Ryder et al., 2006). High flows may reduce algal biomass and shift composition toward early-successional diatoms (Watts et al., 2009a, 2009b). Selective grazing of palatable algal species by invertebrates also may skew biofilm composition (Burns and Walker, 2000a).

DOC and nutrients assimilated into biofilms may short-circuit the path from microbial loop \rightarrow microinvertebrates \rightarrow higher trophic levels, as biofilms may be grazed directly by large macroinvertebrates (e.g. prawns, shrimps, snails) that are prey for fish. Late-successional biofilms tend to be less diverse and have less nutritional value (a higher C:N ratio) than early or mid-stage biofilms. C:N ratios increase between diatoms (Bacillariophyta), non-filamentous and filamentous green algae (Chlorophyta). High C:N ratios (\geq 10:1) in biofilms may reflect dominance of filamentous green algae (Burns and Walker, 2000a). However, Cyanophyta may have comparatively low C:N ratios due to their lack of structural tissue (Burns and Walker, 2000a; Sheldon and Walker, 1997). Algal biofilms appear to be an inadequate food resource for some biota, and may explain the disappearance of snail species from the lower Murray (Wishart, 1994; Sheldon and Walker, 1997; Stevens, 2005).

4.2.2. Influence of regulation

Sheldon and Walker (1997) hypothesized that in the lower Murray reduced water-level variation associated with regulation has caused a shift from heterotrophic to autotrophic dominance of biofilms, and that this has caused a decline in the diversity and abundance of aquatic snails. This is because (i) in turbid systems, small changes in water level affect the light environment experienced by biofilms, (ii) stable water levels reduce the frequency and duration of periods of exposure/inundation and (iii) the velocity of flows within weir pools typically is lower than would have occurred for the same discharge under pre-regulation conditions.

In the mid-Murray, Gawne et al. (2007) calculated that phytoplankton dominate primary productivity (57–68%) and respiration (52–66%), and that the contribution of biofilms contributed <10% of all gross primary production and respiration. Whitworth and Baldwin (2013) suggested that the contribution of biofilms in energy pathways has been reduced by systematic de-snagging programs operating in that region from the mid-1800s until the early 1990s.

4.2.3. Influence of management

Prior to river regulation, levels in the river were characterised by sustained rises and falls that occurred over days-weeks. Under current standard operating regimes, the weirs are managed in response to changes in flow to provide relatively stable upstream water levels (\pm 50 mm). In the weir tailwaters, levels fluctuate daily by about \pm 200 mm (Maheshwari et al., 1995). Management actions that increase the variability may achieve desired ecological outcomes, provided they are within an appropriate time scale.

Potential for re-operation of infrastructure as a control mechanism

Re-operation of existing infrastructure (i.e. weir-pool manipulation) to create more variation in water levels in the Murray weir pools may be used to control biofilm composition by (i) altering the depth of submergence (weir-pool raising) and/or (ii) altering the extent of exposure (weir-pool lowering) of biofilms. The following studies inform such an approach:

- Burns and Walker (2000b) demonstrated that the variation in amplitude and duration of water level in the Lock 1 tailwater during routine weir operations was not sufficient to support maintenance of early-mid successional biofilms in most of the water column in the tailwater or weir pool. They concluded that the magnitude and duration of flows managed for ecological outcomes need to exceed the desiccation tolerances of biofilm organisms.
- Stevens (2005) showed that biofilms with a low C:N ratio (6:1) caused juvenile river snails, *Notopala sublineata hanleyi*, to grow faster than biofilms with higher 10:1 and 14:1 C:N ratios. Biofilms from shallow depths (<0.5 m) displayed elevated C:N ratios (>10:1), and those from 1-m depth had still higher ratios. Alteration of food quality *via* river regulation may have caused the near-extinction of this once common species.
- Souter et al. (2012) investigated the impact of a weir-pool surcharge (mean 430 mm) at Lock 5 from early October to mid-December 2005. Biofilms in this period remained as a mid-successional community for longer than during the control period, but there was no clear effect of increased depth or flow velocity. It was concluded that other factors associated with the pool raising (e.g. nutrients, light, temperature) were implicated.
- Ryder (2004) tested the influence of high, intermediate and low water level variability on biofilms in the Murrumbidgee River (NSW) and demonstrated that biofilms exposed to a high frequency of water level variability where characterised by lower biomass, and were more desirable, being comprised of a heterotrophic biofilm assemblage with an algal community characterised early successional algae compared to biofilms exposed to the other treatments.
- Wallace (2013c) provided evidence that frequent, small-scale changes in depth of inundation can have a significant impact on biofilms in the Lower Darling River (NSW). Biofilms were exposed to four treatments using change in submergence depth from 0.15 m to 0.50 m as a substitute for a 0.35 m increase in water depth over a 50-day period.

Four treatments ranging from stable submergence depth to a weekly switch in inundation depth were tested. Biofilms developed least rapidly in the most variable regime. This is supported by the observation that maximum total biomass, ash free dry mass, and chlorophyll *a* was recorded in the most stable regime. Furthermore, there was a significant difference in community composition between the most variable and the remaining treatments.

Potential for increased flow to alter biofilm composition

The main pathways through which increased flow may alter biofilm composition are outlined below:

- 1. Increased flow will increase the depth of submergence, but to attain maximum benefit the weir-pool height would not be maintained at normal levels (otherwise, any benefit is likely to be limited to the tailwater).
- 2. Water from different sources may alter physico-chemical water quality parameters such as (i) nutrient and carbon concentration and/or bioavailability, or (ii) turbidity.
- 3. Periods of unregulated flow generated by rain events may alter carbon and nutrient concentrations and/or bioavailability or other water quality parameters. For example, unregulated flows from some tributaries (Ovens, Goulburn, Wakool, Murrumbidgee, Darling) may increase turbidity.
- Increased velocity or sediment loads may cause scouring of biofilms. Velocities of ≥0.3 m s⁻¹ can influence biomass and taxonomic composition (Ryder et al., 2006), and suspended sediments cause physical abrasion (Grimm and Fisher, 1989).

Operational constraints/requirements

Studies in the Mitta Mitta River (Vic), summarised by Watts et al. (2009a, b), indicate that the benefits from high-velocity flows are short-lived, with communities reverting to pre-flood communities within days—weeks. In systems held stable for a long period, a large disturbance may be needed to facilitate a return to early-successional biofilms (Burns and Ryder, 2001). Consequently, managing flows intermittently for temporary increases in velocity might not be an effective way to improve biofilm condition in the lower Murray. Raising the height of weir pools during periods of increased flow, to maximise the depth of submergence of biofilms, may be counter-productive. This may occur if the increased velocity that would normally occur with an increase in flow is countered by the increased storage capacity of the weir pool.

Ability to utilise water source to alter the light environment

The Darling River usually is turbid (>100 NTU), and the photic depth is 0.6–0.7 m (Sherman et al., 1998). In contrast, median turbidity in the Lock 10 weir-pool, upstream of the Murray-Darling junction is <25 NTU. Downstream of the junction, median turbidity increases to about 60 NTU (Mackay and Eastburn, 1990). The shallow photic depth (where 1% of incident light remains) associated with high turbidity implies that small changes in water level will substantially alter the light environment for biofilms. In addition to utilising changes in flow or weir-pool operations, the proportions of flow from headwater storages in the Murray system or the Menindee Lakes could be varied to alter the depth of the photic zone

4.3. Harmful and nuisance algal blooms

4.3.1. Background

The phytoplankton is a vital component of aquatic food webs, and algal blooms need not indicate an 'unhealthy' system. Indeed, phytoplankton are an important source of energy in the Murray, as in other rivers (Oliver and Merrick, 2006; Gawne et al., 2007; Oliver and Lorenz, 2007), and the same dependence is evident in the streams of tropical Australia (Douglas et al., 2005). The ideal is for nutrient concentrations to promote both phytoplankton and macrophyte growth, with sufficient water movement to inhibit the development of stratification and hypoxia (Brookes et al., 2009) and avoid a shift towards Cyanobacteria ('blue-green algae').

Blooms are problematic when they impact users of untreated water (i.e. irrigators, graziers) and recreational users and the suitability of water for potable supply (Codd et al., 1994). In the lower Murray, algal blooms have been problematic for 135 years. Scums of algae were reported in 1853, 1878, 1880 and 1888. In 1878, several hundred stock were poisoned at Lake Alexandrina during a bloom of the cyanobacterium *Nodularia spumigena*, during a period of low flow, low lake water levels and high temperatures.

The hazards associated with Cyanobacteria range from public-health toxicity to aesthetic taste and odour issues. The toxins produced by Cyanobacteria include hepatotoxins (liver-damaging) and neurotoxins (nerve-damaging). The taste and odour compounds produced by Cyanobacteria are geosmin (trans-1, 10-dimethyl-trans-9-decalol) and MIB (2-methylisoborneol). These are difficult to remove with conventional methods, and require expensive activated carbon for adequate removal. In addition to impacts to consumptive users (e.g. humans, stock), blooms of Cyanobacteria may affect aquatic food webs. For example, Cyanobacteria are a non-preferred food for invertebrate grazers (Carney and Elser, 1990; De Benardi and Giussani, 1990).

4.3.2. Influence of regulation

Nuisance blooms in the lower Murray typically are dominated by the cyanobacterium *Anabaena circinalis*, and occur during low flows in summer (Burch et al., 1994), controlled by water temperature, light (Photosynthetically Active Radiation, PAR), nutrients, mixing (water-column stability) and 'loss' processes including washout, residence time, sedimentation and grazing (Webster et al., 1997; Maier et al., 1998; Sherman et al., 1998; Baker et al., 2000). Maier et al. (1998) developed an artificial neural network for predicting the growth of *A. circinalis* in the Murray at Morgan. Key variables in the model are flow, temperature and colour, affecting the stratification behaviour of the water column. Concentrations of phosphorus, nitrogen and iron were less important.

Nutrients

See section 4.1.

Retention time

Baker et al. (2000) showed that phytoplankton move laterally/longitudinally at the same rate as the water in the river. Therefore, slow-flowing areas or zones with long retention times may increase algal biomass in the river by allowing extended growth. In addition to phytoplankton cells derived from upstream and in-channel growth, seeding populations may be imported from the floodplain and/or connected wetlands, particularly following rapid drops in water level.

However, localised blooms may not persist for long distances downstream as the river environment generally is not conducive to sustaining blooms (Bormans et al., 1997).

Water column stability

The onset of thermal stratification is a function of flow, solar radiation and wind speed (Bormans et al., 1997; Maier et al., 2001). Persistent stratification can cause a transition from non–motile to buoyant species such as *Anabaena* spp., due to the lack of water-column mixing to entrain non–motile phytoplankton (species that lack buoyancy-control) (Sherman et al., 1998). Hence, low-flow conditions in summer may favour buoyant Cyanobacteria rather than diatoms and other groups. Stratification for 2 days is sufficient for buoyant species such as *A. circinalis* to accumulate in the surface layer, and for short-lived blooms to develop. The magnitude of the problem increases markedly if stratification persists for more than 7 days (Maier et al., 2001).

In addition to issues associated with blooms, thermal stratification is likely to affect dissolved oxygen (DO), plankton and solutes (Becker et al., 2010). Persistent stratification isolates the bottom layer of the water (forming a thermocline and hypolimnion) and prevents oxygen transfer. Respiration and chemical oxygen demand deplete the remaining oxygen, reducing the habitat for fish and other aerobic organisms (Turner and Erskine, 2005).

Persistent stratification can develop in the lower Murray when flow velocities are <0.1 m s⁻¹. During periods of entitlement flow in the lowermost reaches, when water velocity is low (0.04– 0.06 m s^{-1}) and solar radiation is high (as in summer), wind speed is a dominant factor (Bormans et al., 1997; Maier et al., 2001). The depth of the mixed layer can vary from one side of the river to the other due to protection from wind-driven mixing and shading by the riverside cliffs (Bormans et al., 1997).

An assessment of persistent thermal stratification at Nildottie was provided by Burch et al., (2004). This compared low-flow conditions from 16 November 2003 to 8 January 2004 (mean flow downstream of Lock $3 = 4911 \text{ ML day}^{-1}$ with very-low flow conditions from 16 November to 18 December 2007 (mean flow = 1622 ML day⁻¹) (Figures 4.5–4.6). Under very-low flow conditions, stratification was more persistent than under low-flow conditions. Under low-flow conditions, the river exhibited diurnal stratification that did not persist for longer than 3 days. In contrast, under very-low flow conditions, stratification from 2–7 days.

Model of bloom formation

A conceptual model developed by Baker et al. (2000) proposed that during moderate flows in the lower Murray, flow is the dominant factor in development of cyanobacterial blooms. Wind speed and source of water are next most important. The model predicts that:

- At flows >10,000 ML day⁻¹, turbulence favours species like the negatively-buoyant diatom *Aulacoseira granulata*,
- At 10,000 ML day⁻¹, diurnal stratification may occur if wind speed is <1.2 m s⁻¹,
- At 4000 ML day⁻¹, persistent stratification may develop if wind speed is <1.2 m s⁻¹,
- Diurnal stratification may occur at wind speeds of 1.3–3.0 m s⁻¹,
- Wind speeds >3 m s⁻¹ disrupt stratification, irrespective of flow, and

• If water is from the Darling or from upstream flooding, phosphorus concentrations may be elevated and nitrogen concentrations will determine the dominant phytoplankton species.



Figure 4.5. Flows in the period (2002–08) when the incidence of persistent thermal stratification in the Murray at Nildottie was assessed. See text for detail. Data from Burch et al., (2004).



Figure 4.6. Frequency histograms of the duration of thermal stratification in the Murray at Nildottie: [A] low-flow conditions, 16 November 2003–8 January 2004 and [B] very-low flow conditions, 16 November–18 December 2007. Data from Burch et al.,(2004).

4.3.3. Influence of management

Maier et al. (2001) calculated that the probability of thermal stratification persisting for 2 days under entitlement-flow conditions is highest in January (45%) and December (39%). In contrast, probabilities of persistence for 7 days are <1% in January and <2% in December. The recurrence interval for these conditions is 50 years. Under most circumstances, therefore, there is a low likelihood of extensive reaches of the lower Murray being stratified long enough to create a bloom. However, environmental watering may change the risk parameters.

During environmental watering, the release of nutrients from accumulated material on floodplains can produce a nutrient pulse capable of supporting significant phytoplankton biomass (Brookes et al., 2007; Wallace, 2008; Wallace and Lenon, 2010). This could be significant for watering actions that decouple floodplain and river hydrology. Potential examples are:

• Floodplain inundation using constructed infrastructure when discharge is sufficient only to maintain in-channel flow may increase nutrient concentrations and availability,

- During flood recessions using constructed infrastructure, phytoplankton communities in low-flow areas (terminal wetlands or outer floodplain areas) may drain back to the channel,
- Weir-pool raisings to increase stage (water level) and increase the inundated area during periods of entitlement-flow or slightly-elevated flow may reduce flow velocity and/or retention time, and
- Coordinated weir-pool lowering and draining of connected wetlands to the river may entrain increased nutrient loads and/or phytoplankton populations from those wetlands.

Oliver and Lorenz (2013) and Aldridge et al. (2012a) demonstrated large peaks in chlorophyll *a* (85–90 mg m⁻³) prior to and post peak-discharge in the 2010–11 flood. An increase in diatom abundance with increasing discharge and silica concentration was reported by Aldridge et al. (2012a), although diatoms declined once flows exceeded 40,000 ML day⁻¹. Cyanobacterial numbers were highest during low-flow conditions and lowest during the period of highest discharge. The switch from Cyanobacteria to other groups at flows >10,000 ML day⁻¹ is consistent with the model developed by Baker et al. (2000). As conditions during the flood were not conducive to high rates of phytoplankton growth, Oliver and Lorenz (2013) attributed the post-flood peaks to phytoplankton biomass imported from the floodplain.

During periods of low-water availability, river operators and environmental managers may use 'sustaining' flows (Lind et al., 2007) to disrupt thermal stratification and break up cyanobacterial blooms (Maier et al., 2001). Increased flow reduces the probability of persistent stratification, and could be achieved by increasing releases from Lake Victoria. However, the maximum release from Lake Victoria is not sufficient to completely eliminate the potential for stratification. In addition, this is costly: given a cost of \$50 per ML, a release of 10,000 ML day⁻¹ from Lake Victoria for 7 days would cost c. \$5 million. A 300-mm reduction in weir-pool level at Locks 1–3 would have more impact, and may be the least expensive and most effective option.

4.3.4. Environmental thresholds

The environmental trigger for chlorophyll *a* in the ANZECC Water Quality Guidelines (2000) is $\geq 5 \ \mu g L^{-1}$. This is not particularly relevant to the lower Murray, where typical (non-bloom) concentrations are 10–20 $\mu g L^{-1}$ (Oliver and Lorenz, 2013). Values of 5–15 $\mu g L^{-1}$ are normal for lowland rivers such as the Darling and Murray.

4.4. Fish

4.4.1. Background

Flow variability drives fish assemblage structure directly and indirectly. Direct influences impact life-history processes including pre-spawning condition and maturation, spawning cues and spawning behaviour, larval and juvenile survival and migration (Junk et al., 1989; Welcomme, 1985; Humphries et al., 1999; King et al., 2009b). Indirect influences include (i) changes in primary and secondary productivity and (ii) changes in habitat availability *via* inundation of wetlands and floodplains, changing hydraulic conditions, altered channel morphology and the diversity of structural elements, including instream vegetation (Nestler et al., 2012).

For fish populations to be sustainable, the adults need to attain reproductive condition, have access to spawning habitats and spawn successfully. The offspring then need to survive until they reach sexual maturity and breed. Delivering environmental water allocations to promote spawning therefore will not necessarily increase fish populations; it is necessary to provide habitat and resources to support all stages of the life cycle (King et al., 2009a).

4.4.2. Influence of regulation

Throughout the Murray-Darling Basin, the abundance and distribution of native fish have declined (Cadwallader, 1978; Harris and Gehrke, 1997; MDBC, 2004) to approximately 10% of pre-European levels (MDBC, 2004). Numerous authors have suggested that river regulation is the main cause of the decline (e.g. Walker, 1985; Cadwallader and Lawrence, 1990; Gehrke et al., 1995). Associated factors include the construction of barriers to movement, altered temperature regimes, reductions in aquatic vegetation and deeper pool habitats, loss of reproductive cues and reduced floodplain access (King et al., 2010). Balcombe et al. (2011) concluded that the largely unregulated rivers of the upper Murray-Darling Basin are in better ecological condition, and better able to support recruiting populations, than regulated rivers where reduced flood frequency, intensity and duration have suppressed recruitment due to successive years without strong flood pulses. Gehrke et al. (1995) concluded that regulation may alter the relative abundance of native fishes and introduced fishes by desynchronising environmental and reproductive cycles.

4.4.3. Influence of management

Recruitment models and spawning modes

To predict the likely response to a given flow, it is necessary to understand the spawning modes and recruitment models for the species present. Two models have been proposed for temperate Australian floodplain rivers:

- The *Flood-Recruitment Model* (Harris and Gehrke, 1994) proposes that a rising hydrograph cues spawning in some species, and that floodplain inundation provides increased productivity and suitable habitat for fish larvae and early juveniles, and
- The *Low-Flow Recruitment Model* (Humphries et al., 1999) proposes that the falling-limb and low-flow components of the hydrograph provide most opportunities for recruitment success, particularly for small-bodied species.

Humphries et al. (1999) outlined three spawning modes:

- *Mode 1 species* (e.g. Murray cod) spawn around the same time every year, initiated by a combination of circa–annual rhythms and temperature, and do not require an increase in discharge to trigger spawning. However, increases in flow and flooding may improve larval survivorship and recruitment (Ye et al., 2000).
- Mode 2 species (i.e. golden perch, silver perch) are flood/flow cued spawners, requiring increases in discharge to initiate spawning and promote larval survivorship. Several studies include common carp in this mode (King et al., 2003a; Stuart and Jones, 2006), although there is also evidence of annual recruitment irrespective of flow conditions (Smith and Walker, 2004).

• *Mode 3a* (e.g. flat-headed gudgeons) and *3b species* (e.g. Murray rainbowfish) will spawn and recruit under a range of flow conditions. Although none is cued by changes in flow, many species are abundant during low-flow conditions.

Kirby et al. (2013) concluded that both of the recruitment models and the three spawning modes are applicable within the lower Murray. King et al. (2009a) observed that many Murray-Darling species have flexible spawning periods and timing can vary across years and rivers, although some, like Murray cod, spawn annually over well-defined seasons. Further, low-flow periods may be important for some species (mode 3 spawners), whilst elevated flows and/or flood are required for spawning and recruitment of others (mode 2 spawners) (Humphries et al., 1999). It is unlikely that a narrow range of flow pulses (single, micro-scale events) would benefit all species (King et al., 2010).

Magnitude of flow to elicit a response

Baumgartner et al. (2013) identified three 'groups' of species in the context of flow responses:

Long-lived apex predators — Murray cod

Murray cod occur in a variety of aquatic environments, favouring rivers, anabranches and creeks rather than wetlands and temporary floodplain habitats (King, 2004; Koehn, 2009; Leigh and Zampatti, 2013). The species is long-lived (up to 60 years) and annual recruitment therefore is not necessary to maintain a sustainable population. Spawning occurs in spring to early summer regardless of flow conditions, but in response to photoperiod and water temperature (Rowland, 1998; Humphries, 2005; Koehn and Harrington, 2006). Nonetheless, Baumgartner et al. (2013) suggest that environmental flows could enhance spawning by inundating more potential spawning sites and providing opportunities for juveniles to visit off-channel nursery habitats. This is based on data from the deeply-incised Darling River (Boys, 2007), where modest increases in flow result in significantly increased stage. It is unlikely in the lower Murray, due to its greater width and low bank slope. Whilst annual spawning is observed in the lower Murray (Zampatti et al., 2010; Ye et al., 2013), strong recruitment is observed only in years following elevated flows (Ye and Zampatti, 2007). Assessments of population demography suggest that there is limited recruitment from spawning in low-flow years, but strong cohorts have resulted from years of high flow (>40,000 ML day⁻¹) (Ye and Zampatti, 2007). The causal link between elevated flow and recruitment of Murray cod in the lower Murray is unknown, but larvae drift downstream following hatching, and this would be supported by elevated flows. Additionally, Leigh and Zampatti (2013) recorded large-scale movements during high flows.

Flow dependent specialists – golden perch and silver perch

Golden perch have been extensively studied due to their recreational and commercial importance. River regulation and the associated disruption of upstream and downstream movement caused by reduced lateral and longitudinal connectivity have acute impacts on species that migrate and utilise pelagic spawning strategies (Dudley and Platania, 2007). Golden perch have a flexible life history and will spawn and recruit in conjunction with rises in flow in the channel and overbank flows (Zampatti and Leigh, 2013a). Recruitment is minimal during stable low flows (Ye et al., 2008; Zampatti and Leigh, 2013b). Golden perch are long-lived (20–25 years) and populations do not need to recruit every year. However, the types of flow events that

have been correlated with spawning post-regulation (i.e. seasonal in-channel flow pulses) would have occurred on a near-annual basis prior to regulation (Zampatti and Leigh, 2013b).

Zampatti and Leigh (2013a) reported that during 2005–09 the age structure of golden perch in the Chowilla region was dominated by four cohorts spawned in 1996 (overbank flow), 1998 (inchannel moderate flow), 2000 (overbank flow) and 2005 (in-channel flow spike). In 2010, a cohort of young-of-year fish was recorded. In 2011, the demographic was dominated by fish spawned in 2009–10 (54%) and 2010–11 (29%). The remaining demographic was comprised of 5-, 10- and 12-year old fish. Abundance was significantly greater in 2010–11 than in the preceding six years, apparently due to numbers of fish spawned during the 2009–10 in-channel flows and the 2010–11 overbank flood. It was suggested that the cohort spawned in 2009–10 did not originate from the Murray (which was experiencing low flows: 5000–10,000 ML day⁻¹), but following a rise in discharge in the lower Darling, and that juveniles colonised the lower Murray by larval drift and/or downstream dispersal.

Foraging generalists

Most foraging generalists are relatively short-lived (i.e. Australian smelt, unspecked hardyhead (< 3 years), Murray rainbowfish, carp gudgeons, flatheaded gudgeons (<5 years) (Baumgartner et al., 2013). Life expectancy for bony herring is 5–8 years (Puckridge and Walker, 1990). Note that flatheaded gudgeon(s) and Murray rainbowfish, defined by Baumgartner et al. (2013) as foraging generalists, were classified by Humphries et al. (1999) as Mode 3a and 3b species, respectively.

These species are common in both channel and wetland habitats. Low-flow periods are conducive for recruitment and elevated abundance, and high flows appear to render channel habitats unfavourable due to the loss of submerged macrophyte cover (Bice et al., 2013). Lateral connection to off-channel habitats is likely to be important, especially during high flows when channel habitats are unfavourable

Many of these species have flexible spawning and recruitment strategies. Several species (e.g. Australian smelt) spawn multiple times over a protracted season, while others spawn once, or a few times, over a more-defined season (e.g. Murray rainbowfish). Elevated flow is not required to stimulate spawning or enhance recruitment, as high abundances of larvae and subsequent recruitment are associated with periods of low flow (Cheshire et al. 2012, Bice et al., 2013). Nonetheless, numerous small-scale events, with small rises in stage to inundate spawning habitat, may be advantageous. Given that many of these species have sticky demersal eggs, inundation would need to persist long enough for eggs to develop and hatch (<10 days) and occur frequently enough to allow 2–3 breeding opportunities within the life span. Baumgartner et al. (2013) suggested that flow management for these species should focus on habitat availability and lateral connectivity rather than large-scale events.

Floodplain specialists

Baumgartner et al. (2013) also defined southern purple-spotted gudgeon (lifespan <10 years) and Murray hardyhead (<3 years) as floodplain (wetland) specialists, as most life-cycle stages occur in wetlands (Ralph et al., 2010). Both species are present in the lower Murray at a few spatially-fragmented sites (in the case of southern purple-spotted gudgeon, only one wetland), reflected in their conservation status and protection under federal (*Environment Protection and*

Biodiversity Conservation Act 1999, Murray hardyhead) and state legislation (*Fisheries Management Act* 2007, southern purple-spotted gudgeon). Given their limited distribution and low abundance, data on their response to flow are limited.

The ecology and distribution of Murray hardyhead are described by Wedderburn et al. (2007; 2008a, b). The species is found in wetlands (both permanently and temporarily connected) with salinity regimes influenced by irrigation run-off and exchange with the main channel. Thus, the species may respond to small, site-scale watering events, particularly in disconnected wetlands, but also to large-scale events. Given that these species have sticky, demersal eggs, inundation would need to persist long enough for eggs to develop and hatch (<10 days) and occur frequently enough to allow 2–3 breeding opportunities within their life span. Baumgartner et al. (2013) suggested that flow management for these species should focus on habitat availability (including favourable physico-chemical conditions) and lateral connectivity rather than large-scale events. Nonetheless, high-flow events probably were an important feature of the hydrograph for Murray hardyhead, allowing dispersal to newly inundated floodplain habitats.

Other (freshwater catfish)

Freshwater catfish were historically common in the lower Murray, supporting a commercial fishery with annual catches of >20 tonnes in the late 1960s and early 1970s (Ye and Hammer, 2009), but their distribution now is limited to a few wetlands and the main channel (Hammer et al., 2009). They make limited short-distance movements, deemed unrelated to spawning (Davis, 1977; Merrick and Midgley, 1981; Reynolds, 1983). The life span is 8-12 years (Clunie and Koehn, 2001), so that annual recruitment is not vital to maintain a sustainable population; conversely, recruitment failure in consecutive years can have a detrimental impact on local populations. Freshwater catfish is a nester and flow-independent spawner (Bice, 2010), and water temperature (>20°C) is the primary stimulus for spawning (Pusey et al. 2004). Eggs are spawned into a nest of pebbles, gravel and other coarse material; eggs are non-adhesive and are guarded by the male (Davis, 1977). Spawning is from October to March, with peak activity from October–January (indicated by the presence of larvae) in the lower Murray (Ye et al., 2013). Increased flow may enhance recruitment, but fast recession of water levels may result in exposure and abandonment of nests. Baumgartner et al. (2013) suggested that flow management for freshwater catfish should focus on habitat and food resources availability as well as timing of the flow. Nevertheless, more specific knowledge of the flow and habitat requirements of this species in the lower Murray is needed to inform flow management.

Timing

Many species may have temperature thresholds below which spawning will not occur. Low temperatures may also cause mortality of eggs in some species such as Murray cod (Ryan et al. 2003, cited by Koehn and Harrington, 2006). In a study at Lindsay Island (an anabranch of the lower Murray in Victoria), Vilizzi (2012) reported peak abundance for native fish larvae in September–February/March, when water temperatures are >15°C. A 'spawning calendar' is shown in Table 9.1, outlining (i) minimum temperatures and (ii) the months in which larval fish have been detected in the lower Murray. The calendar shows that none of the large-bodied native fish are likely to spawn if temperatures are <16°C, and that golden perch and silver perch require temperatures >20°C. However, silver perch may spawn in the mid–upper Murray at temperatures of 16-17°C (King et al., 2013).

An example of the importance of timing of flow relative to temperature is provided by Zampatti and Leigh (2013b), in that spawning and recruitment of golden perch were observed following a discharge of 15,000 ML day⁻¹ in spring 2005, but not following an event of the same magnitude in late winter 2003. A similar pattern was observed by Vilizzi (2012).

Flooding in late winter—early spring, in conjunction with low temperature, may not initiate spawning or stimulate primary and secondary productivity, leading to a lack of food for larvae (Humphries et al., 1999; Roberts et al., 2008). This knowledge, combined with an understanding of spawning modes, is needed to identify:

- The environmental factors (photoperiod, moon phase, flow components, water temperature) that cue reproductive effort (e.g. gonadal development, migration) and spawning, and
- The conditions required for development and survival through egg, larval and early juvenile stages.

Identifying these requirements, and managing the environment appropriately, are important as spawning is a precursor for recruitment, and survival and growth during early life stages will largely determine recruitment success and cohort strength. King et al. (2009a) reported that, for most species, more is known of the flow and habitat requirements for spawning than the conditions required by larvae and for recruitment.

Lateral connectivity and floodplain inundation

Whilst *lateral connectivity* can encompass all connections between the channel and off-channel habitats, including temporarily inundated floodplain habitats, it is important to distinguish 1) lateral connectivity between the main channel and wetlands that are inundated at 'pool level' or under low-moderate flows and 2) lateral connectivity between the channel and temporarily inundated floodplain habitats during flooding (>40,000 ML day⁻¹).

Lateral connectivity under low-moderate flows (10,000–40,000 ML day⁻¹) is important for foraging generalists. These species are commonly abundant in main and wetland habitats and movements between these habitats are documented for the upper (Lyon et al., 2010) and lower Murray (Conallin et al., 2011). Their purpose is unclear, but feeding, spawning and recruitment may occur in both habitats.

Few native species utilise temporarily-inundated wetlands as spawning habitat (Humphries et al., 1999; King et al., 2003b). Indeed, apex predators (Murray cod) and flow-dependent specialists typically spawn and recruit within the channel. An increase in discharge and temperature may provide golden perch with cues for upstream migration and spawning (Leigh and Zampatti, 2011) and downstream dispersal of larvae and young-of-year fish (Humphries, 2005; Zampatti and Leigh, 2013a). Further, channels provide habitat to support juvenile golden perch (Mallen-Cooper and Stuart, 2003; Zampatti and Leigh, 2013b). This does not preclude the importance of floodplain inundation for native fish. Young-of-year golden perch and other species were recorded in high abundance in ephemeral floodplain habitats in the lower Murray following the 2010–11 floods (B. Zampatti, unpublished). Juvenile golden perch were also found to actively move between the channel and connected wetlands under low-moderate flows in 2012-13 (Earl and Ye, unpublished).

Ye et al. (2008) proposed that strong year-classes of golden perch in 1996–97 and 2000–01 demonstrate the importance of high flow/flooding for recruitment in this species. They suggested that the relationship is driven by enhanced spawning and enhanced survival of larvae and early juveniles. Cheshire et al. (2012) demonstrated enhanced spawning of golden perch (indicated by larval abundance) in the lower Murray during the 2010–11 flood, compared to previous low-flow years. Zampatti and Leigh (2013a) concluded that there is a clear association between flooding and recruitment for golden perch, and proposed that overbank flows facilitate local spawning and recruitment and/or redistribute larvae and juveniles from upstream. However, they added that intermediate flows that remain in-channel, between large overbank flows, may be no less important, supporting more frequent spawning and recruitment and therefore more robust native populations. This is supported by other studies indicating strong recruitment associated with in-channel flow pulses and overbank flows (Mallen-Cooper and Stuart, 2003; Zampatti and Leigh, 2013b).

Differences between valley and gorge reaches

Ye et al. (2008) recorded juvenile golden perch in 2006–07 and 2007–08 that were spawned during the 2005–06 in-channel flow 'spike'. However, most of these fish were recorded in the valley (floodplain, anabranches) rather than the gorge reach. Further, the size structure differed between these reaches. The gorge population was smaller, attributed to slower growth rates. A similar observation was reported by Zampatti and Leigh (2013a), where (i) the size structure of golden perch differed between valley and gorge reaches, and (ii) a cohort of young fish was recorded in the valley but not the gorge reach. They suggested that the effects of weirs on upstream and downstream movement of larval, juvenile and adult fish may be responsible. Ye et al. (2008) also reported differences in the size structure of the populations of bony herring (*Nematalosa erebi*) and common carp (*Cyprinus carpio*) between the valley and gorge reaches.

It is possible that the length of available downstream habitat may influence o differences in fish communities between the valley and gorge reaches. Anderson et al. (2006) proposed that for riverine species that disperse downstream there is a minimum habitat length below which production cannot balance downstream losses. Larvae and early juveniles that drift from the lower gorge reach to the lower lakes enter an environment dominated by open-water zones that may not be conducive to recruitment in those species.

Possible flow scenarios

King et al. (2003b) proposed that for an EWA to promote spawning and recruitment, the hydrograph would need to (i) couple high flows and temperatures, (ii) be within a predictable time phase for that system, (iii) have slow rates of rise and fall, (iv) have a duration of several weeks and (v) inundate a large proportion of floodplain. Mallen-Cooper et al. (2008) concluded that it is more beneficial to deliver more flow infrequently than less flow frequently. Based on back-calculated spawning dates of golden perch that indicate spawning on both the rising and falling limbs of hydrographs, and drawing from other studies, Zampatti and Leigh (2013b) proposed trialling delivery of environmental water with some variation in the magnitude of the peak (e.g. two sequential peaks \geq 15,000 ML day⁻¹) when temperatures are above the threshold (c. 20°C) in late spring–early summer.

Table 4.2 A 'spawning calendar' for native and non-native fish. Data are for studies at Chowilla, Lindsay Island and in the Lower Murray Region (Darling to Lower Lakes). Grey bars represent detection of larval stages. Data from King et al. (2009a). Note that there may be regional variation in thresholds and timing (for example, silver perch may spawn at temperatures of 16–17°C: King et al., 2013).

Minimum				Month										
	Species	(°C)	Jul	Aug	Sep	Oct	Nov	Dec	Jan	Feb	Mar	Apr	May	Jun
odied native	golden perch	22												
	silver perch	22 [16-17]												
	Murray cod	17												
ge-bo	freshwater catfish	21												
Larg	bony herring	16												
died native	Australian smelt	15												
	flatheaded gudgeons spp.	15												
	carp gudgeons spp.	15												
all-bo	unspecked hardyhead	17												
Sm	Murray-Darling rainbow fish	17												
Non-native	common carp	15												
	redfin perch	16												
	eastern gambusia	20												
	goldfish	15												



Figure 4.7. Mean seasonal water temperature in the River Murray upstream of Customs House (DEWNR station A4261022) from July 2002 to June 2013. Error bars are ±1 standard deviation. The intersection of the (i) broken and (ii) solid, vertical and horizontal reference lines depicts the time in spring when water temperatures increase to 15°C and 20°C, respectively.

Small, frequent variations in water level and hydraulic complexity may benefit foraging generalists and floodplain specialists. Reproductive activity will occur under flows in the range 15,000–20,000 ML day⁻¹, given appropriate temperatures. However, higher (bankfull) flows that expand lateral connectivity may provide more habitat and resources needed to achieve substantial recruitment. Not all fish will benefit to the same degree each year, as each group has different flow requirements. A variety of flows over appropriate time scales is required to achieve a 'sustainable fish community' (Baumgartner et al., 2013).

4.5. Aquatic and understorey vegetation

4.5.1. Background

Within the SA Murray Corridor, there are 184 native species found in the aquatic or floodplain environments that do not occur in adjacent terrestrial ecosystems (Jessop and Tolken, 1986; Cunningham et al., 1992; Jessop et al., 2006. The habitats provided by these species are unique. Some of the roles of aquatic plants include primary production, physical habitat for fish, birds and invertebrates, improved water quality and erosion control (Kirby et al., 2013).

The main factors that influence plant recruitment in wetland and floodplain habitats are the presence or absence of standing water (van der Valk, 1981b), depth, duration, frequency and timing of flooding (Casanova and Brock, 2000), soil salinity (Brock et al., 2005a; Nielsen et al., 2003b) and soil moisture (Nicol, 2004). Native flood-dependent and flood-responsive plants are typically short-lived, with flooding required to remove drought-tolerant species that become established during the drying phase (Nicol, 2004). Aquatic macrophytes have propagules that

allow them to establish while the substrate is flooded (van der Valk, 1981b), but the germination and establishment of most amphibious species occur in shallow water or when water levels are receding (van der Valk, 1981a; Brock and Casanova, 1997b; Casanova and Brock, 2000; Bell and Clarke, 2004). This is likely to be because in shallow water (0–20 cm) the availability of water and atmospheric gases is optimal (Blanch et al., 2000a). Further, synchronising germination with receding water levels allows plants to take advantage of bare substrate with high soil moisture (Capon, 2003; Nicol, 2004; Capon and Brock, 2006).

Roberts et al. (2000) suggested that 'water regime' should be considered as a time series of three states: *inundated and submerged*, *inundated but not submerged* and *not inundated*. Key statistics for each are mean duration, variability of duration, variability of the period between the occurrence of one state or another and seasonal occurrence of the three states. During a period of frequent flooding, the plant community is dominated by amphibious and flood-dependent species for approximately 12 months and is then progressively replaced by drought-tolerant species until the next flood. In wetlands that are inundated on a near-annual basis, the shift towards terrestrial species is not likely to be observed.

Kirby et al. (2013) suggested that autotrophic production from vegetation that develops on the recession of inundation events is more likely to contribute to the terrestrial food web than to the aquatic food web, because most of the species that respond positively to flooding have completed their life cycle and senesced prior to the next flood. However, the biomass of species tolerant to dry conditions provides a large source of carbon and nutrients to fuel ecosystem metabolism during the next inundation phase (section 5).

Existing conceptual models

Species can be categorised into functional groups based on water-regime preferences and water requirements (Brock and Casanova, 1997a) (Table 10.1).

Nicol et al. (2010a) proposed a state- and-transition model for floodplain vegetation, including four states:

1. Flood-dependent and amphibious vegetation

Areas of floodplain that were inundated in the previous 12 months or the pool-level littoral zone with permanently high soil moisture; amphibious and flood-dependent species are dominant taxa. If the floodplain is not inundated within the 12-month period, the flood-dependent and amphibious taxa will be replaced progressively by drought-tolerant terrestrial species (state 2).

2. Terrestrial dry species

When an area of floodplain dominated by terrestrial species is inundated, the terrestrial species will be replaced by amphibious and flood-dependent taxa. If the area is inundated within 12 months, it will return to state 1. If the area is (i) not inundated and (ii) overlies a shallow saline water table, salt will accumulate in the root zone and the terrestrial species will be replaced by salt-tolerant species (state 3), or the soil salinity may become so elevated that nothing can survive (state 4).

3. Salt-tolerant species

4. Bare soil

When the floodplain is dominated by salt-tolerant species or bare soil, inundation may not result in the return of flood-dependent or amphibious species without multiple watering. However, there is evidence from surveys conducted after the 2010–11 flood that this model may not hold for all areas (Gehrig et al., 2012), and that hydrochory may influence outcomes.

4.5.2. Influence of regulation

Changes to hydrology alter the habitat template, describing the physical, chemical and biological character of rivers and wetlands (Walker and Thoms, 1993). In unregulated systems, during a given flow scenario, a river system will follow a dynamic of dry, drying, lotic and lentic habitats. Regulation has had consequences for plant communities, as hydrology coupled with geomorphology (elevation) governs plant communities in aquatic and floodplain systems (Mitsch and Gosselink, 1993), such that habitats are more frequently either wet or dry. The extent of 'drying' habitat is reduced, hence plant diversity is reduced. It should be noted that prolonged or permanent inundation also can lead to low diversity (Barrett et al., 2010; Nielsen and Chick, 1997; van der Valk, 1994), and in the extended absence of floods, drought-tolerant species become dominant (Nicol, 2010a).

Changes to the timing of flows interact with life-history adaptations, and changes to inundation depth and duration interact with morphological and physiological adaptations (Lytle and Poff, 2004). Vegetation surveys by Blanch et al. (2000a) in the River Murray during 1994 followed overbank floods in 1992–93, and recorded a 'cone-shaped' longitudinal distribution that varied according to location within the weir pools. In the upper 10% of a weir pool, most species occurred in a band up to 4–6 m above pool level. In the lower 10% of the pool, where water levels are most stable, most species occurred in a band up to 1–1.5 m above pool level. Species adapted to stable water levels, such as *Typha domingensis* and *Schoenoplectus validus*, were restricted to the lower parts of the weir pools.

Water-level stabilisation has reduced the elevation bands at which water regimes occur, so that plants with differing water requirements are constrained to narrow bands where those requirements are met. During the Millennium Drought, extreme stabilisation of water levels resulting from low water availability and tightly regulated flows narrowed the littoral zones available for colonisation by amphibious and floodplain species to a strip no more than 1-m wide (Marsland and Nicol, 2008; Nicol et al., 2010a; Nicol et al., 2006). Species recorded by Blanch et al. (2000a) as confined to lower weir pools extended their distribution into the mid-reaches (Marsland et al., 2010), and species in the main channel were dominated by emergent species including willows (Salix spp.), cumbungi (Typha spp.), common reed (Phragmites australis) and club rush (Schoenoplectus validus). Submergent species including ribbon weed (Vallisneria australis), curly pondweed (Potamogeton crispus) and milfoils (Myriophyllum spp.) were restricted to shallow areas (Nicol et al., 2010a). Surveys after the 2010–11 flood reported species adapted to wetting and drying more than 2m above normal pool levels (Gehrig et al., 2012). Low turbidity, deep photic zones and low velocity during periods of low water availability and flows from storages in the upper Murray greatly influence the abundance and distribution of submerged species in the lower channel (Nicol et al., 2006; Marsland and Nicol, 2008; Marsland et al., 2010).

Table 4.3. Plant functional	groups. Adapted from	n Brock and Casanova (1997).

Functional Group	Conditions
Terrestrial dry	Intolerant of flooding and will persist in environments with low soil moisture
Terrestrial damp	Tolerate inundation for short periods and require high soil moisture to complete their life cycle.
Floodplain	Intolerant of long-term inundation. Restricted to areas that flood periodically because they do not germinate in response to rainfall. Germinate after flood waters recede or wetlands are drawn down.
Amphibious fluctuation tolerator– emergent	Mainly emergent sedges and rushes that prefer high soil moisture or shallow water but require their photosynthetic parts to be emergent. May tolerate short-term submergence.
Amphibious fluctuation tolerator– woody	Woody perennial species. Similar water regime preferences to the preceding group. Several species are tolerant of desiccation for extended periods.
Amphibious fluctuation tolerator– low growing	Similar water regime preferences to the preceding two groups. Some can grow totally submerged except during flowering.
Amphibious fluctuation responder–plastic	Respond to water-level changes <i>via</i> rapid shoot elongation or a change in leaf type. Can persist on damp and drying ground because of morphological flexibility, and can flower even if site does not dry out.
Floating	Float on the surface (often unattached to sediment) with most species requiring the presence of free water year-round. Some species can survive and complete their life cycle on mud.
Emergent	Require permanent shallow water or a saturated root zone, but have emergent leaves or stems.
Submergent <i>r</i> -selected	Colonise recently-flooded areas. Many require drying to stimulate germination and have short life cycles.
Submergent K–selected	Species require permanent water >10 cm deep for more than a year to either germinate or reach sufficient biomass to start reproducing.

4.5.3. Influence of management

Increased variability in hydrology (depth and discharge) is likely to increase the width of the littoral zone and increase plant diversity. Surveys at wetlands where environmental water allocations have been delivered and unregulated flooding has occurred demonstrate an increase in amphibious and floodplain species and a corresponding decrease in terrestrial species and areas of bare soil (Gehrig et al., 2012; Nicol et al., 2010b) even in areas that had not been inundated for over ten years (Gehrig et al., 2012; Holland et al., 2013).

Vegetation surveys after a weir raising to extend a small environmental flow (32 GL day⁻¹) during the drought showed that flood-tolerant and flood-dependent species (e.g. *Sporobolus mitchelli*) grew and germinated, flood-intolerant species (e.g. *Atriplex vesicaria*) senesced, and that no aquatic plants germinated or established (Siebentritt et al., 2004). The authors suggested that the value of short-term environmental flows was to maintain existing communities rather than to restore degraded communities.

4.5.4. Environmental thresholds

Return frequency

Nicol et al. (2010a) suggested that a return frequency of 3-5 years is sufficient to maintain seed banks and ensure the long-term persistence of amphibious and floodplain species. This is supported by Brock (2011), who demonstrated that viable seed for more than 70% of the species originally present in wetland soil cores survived drought conditions for longer than 5 years. This indicates that soil seed banks are highly resilient. However, the risk of loss of taxa during long droughts is real. Once the soil seed bank of flood-dependent and amphibious species has been depleted, a single flood event is likely to produce a muted response (*sensu* Nicol et al., 2010a). Re-establishment will be heavily dependent on the distribution of propagules elsewhere in the landscape and factors (flow direction and connectivity, wind and biota) that facilitate dispersal (Green et al., 2008; Questad and Foster, 2008).

Timing

Monitoring of EWAs near Mildura (Vic) has demonstrated that autumn-winter flooding does not produce as strong an ecological response as water delivered in spring-summer (Chatfield, 2007; D'Santos, 2007). Further, Campbell et al. (unpublished) have demonstrated a seasonal difference between the effects of spring and winter floods on understorey composition, using soil cores from a wetland at Lindsay Island (Vic). In addition, seasonal differences in exotic species have been observed in wetlands where EWAs have been delivered (Nicol et al., 2010b).

Duration and timing of flooding can affect the capacity of macrophytes to reproduce as (i) standing water needs to persist long enough for life cycles to be completed, if new propagules are to be added, and (ii) growth and reproduction vary seasonally (Warwick and Brock, 2003).

Rates of rise and fall

For the establishment of flood-dependent plant species, rates of rise are not generally important as many will not germinate until water levels are drawn down and the soil is exposed to the air but still retains a high moisture content (Nicol, 2004). Established stands of low-growing emergent amphibious macrophytes may be more vulnerable to rapid increases in water depth than submerged and free-floating aquatic species. Many of these species are unable to maintain photosynthesis and gas exchange to survive extended periods of inundation (Siebentritt and Ganf, 2000). Some emergent macrophytes can grow taller to maintain photosynthetic leaf or stem tissue above the water surface, but this can be detrimental to long-term survival and future vegetative regeneration capacity if it occurs at the expense of below-ground biomass (e.g. rhizomes, tubers) (Rea and Ganf, 1994;Blanch et al., 1999a; Siebentritt and Ganf, 2000).

Soil salinity

Where dissolved salt concentrations are high, water uptake by plants can be restricted (SKM, 2010b). Soils may be defined as saline when the concentration of salts in the soil water is sufficient to cause reduced plant growth (Rengasamy et al., 2009). Conditions where the total soil moisture pressure exceeds the physiological capacity of a plant's root membrane to extract water are referred to as 'osmotic drought' (Overton and Doody 2006). The effect of saline soils is similar to drought conditions in non-saline soils, as the ability of plants to extract water is reduced. The effect is compounded as the soil dries between floods and rainfall events as the salinity increases even though the salt content may remain static (Rengasamy et al., 2009).

Based on observations that *Atriplex* spp. generally do not occur at soil salinities > 20,000 μ S cm⁻¹, this may be the point at which a switch from drought- to salt-tolerant species occurs (Hassam, 2007). However, soil cores from Pike Floodplain (Wallace and Rengasamy, 2011) and Chowilla Floodplain had median EC values <15,000 μ S cm⁻¹ (Wallace, 2013a), suggesting that a threshold of 20,000 μ S cm⁻¹ may be too high.

4.5.5. Flow-inundation curves

Flow-inundation curves (Figure 4.8) demonstrate that the area of forbland inundated at 20,000 ML day⁻¹ is 8 ha (0.34% of the total), and does not increase until flows reach 40,000 ML day⁻¹, when 10.9 ha (0.46% of the total area) is inundated. There is also a slow rate of response for sedgeland. However, at flows of 20,000 ML day⁻¹, 47% of the total area of sedgeland is inundated, and at 40,000 ML day⁻¹, 56% of sedgeland is inundated. The rate of response is relatively consistent for grassland areas. At flows of 20,000 ML day⁻¹, 214 ha (6.6% of the total area) of grassland is inundated, and at 40,000 ML day⁻¹. 513 ha (15.7% of the total area) of grassland is inundated. Shrubland shows that strongest response, with a step-change in area inundated between 35,000 and 40,000 ML day⁻¹. At flows of 20,000 ML day⁻¹, 478 ha is inundated (2.6% of total area); at 35,000 ML day⁻¹ this increases to 747 ha (4.1% of total area), and at 40,000 ML day⁻¹ it increases to 1230 ha (6.7% of total area).



Figure 4.8. Flow-area inundation plots for floodplain vegetation. Note the different axes for A–B.

4.6. Lignum

4.6.1. Background

Tangled lignum (*Duma florulenta*) is a native, woody, usually leafless, multi-stemmed perennial shrub up to 2–3 m high and 3-m diameter, and often forms dense thickets in depressions on floodplains of the Murray-Darling Basin (Sainty and Jacobs, 1981; Cunningham et al., 1992; Jensen et al., 2008c). Lignum provides structural habitat for terrestrial animals during dry periods and aquatic biota, such as fish and invertebrates, during floods (Young 2001). When inundated, lignum shrublands provide breeding habitat for colonial waterbirds such as the straw-necked ibis, sacred ibis and glossy ibis (Lowe 1982; Maher and Braithwaite, 1992). Flooded lignum stands are also used for nesting by the endangered freckled duck (*Stictonetta naevosa*) (Frith, 1965; Lowe and Lowe, 1974; NRE, 2000a, 2000b). Lignum is commonly associated with cracking grey clay soils and usually occurs in ephemeral swamps, shallow flood-runners (Sainty

and Jacobs, 1981; Roberts and Marston, 2000) and areas of flat-sloping (watershed) floodplain (SKM and Roberts, 2003).

Lignum plants often appear lifeless, but endure dry conditions by means of a persistent lignotuber, up to 3 m deep, that maintains contact with subsurface water (Craig et al., 1991; Chong and Walker, 2005). Lignum requires flooding every 3-10 years (possibly more frequently in saline soils) for periods of up to 12 months, and is intolerant to either sustained dry periods or prolonged flooding (Craig et al., 1991). Above-ground biomass of lignum often appears in poor condition (dry, brown, leafless stems) during extended dry periods, but it regenerates a flush of green shoots, leaves and flowers within two weeks of rainfall or flooding.

4.6.2. Flow-inundation curves

Of the 11,709 ha of lignum communities that have been mapped in the SA Murray floodplain, 75% is in the valley reach (Border to Overland Corner), and the remaining 24% is in the gorge reach (Overland Corner to Mannum). Less than 1% is in the lower Murray swamps reach (Mannum to Wellington). Flow-area inundation plots (Figure 4.9) demonstrate that for flows of 20,000-40,000 ML day⁻¹ the response curve is slow. The regression line in Figure 4.9B indicates an exponential response, but with a low rate constant:

$$y = 27.61 + 0.01855 e^{0.000248x} (r^2 = 0.9957, P = 0.0041)$$

At 20,000 ML day⁻¹, 271 ha (2.3% of total) is inundated. With an increase from 20,000 to 35,000 ML day⁻¹, an additional 107 ha (+0.92% of total area) is inundated. For an increase from 20,000 to 40,000 ML day⁻¹, an additional 391 ha (+3.3%) is inundated.



Figure 4.9. Flow-area inundation plots for lignum. The regression in B indicates an exponential response: $y = 27.61 + 0.01855 e^{0.000248x}$ ($r^2 = 0.9957$, P = 0.0041).

4.6.3. In-channel Environmental Watering Requirements

Given the low flow-inundation area responses, in-channel EWRs are not provided for lignum.

4.7. Floodplain eucalypts

4.7.1. Background

River red gum (*Eucalyptus camaldulensis*) and black box (*E. largiflorens*) are responsive to groundwater depth and salinity (Overton and Jolly, 2004). In areas of low rainfall, high potential evaporation and high groundwater salinity, trees extract water from the upper soil profile and are sensitive to salt accumulation during dry periods (Jolly et al., 1993). These trees rely on periodic flooding to improve soil-water availability. Responses to flooding (new growth) are typically due to increased soil moisture in the upper soil profile and not necessarily to changes in salinity in the deeper profile (Overton and Doody, 2008). Where groundwater conductivity exceeds $30,000 \ \mu S \ cm^{-1}$ and $55,000 \ \mu S \ cm^{-1}$ for river red gum and black box, respectively, and depth to groundwater is shallow, tree health is likely to decline (SKM, 2010b). As flooding frequency increases, critical depth to groundwater decreases. The critical depth in infrequently flooded (>100,000 ML day⁻¹) black box communities is 4 m. In areas flooded occasionally, the critical depth decreases to 2–3 m due to flushing of the soil profile (Overton and Jolly, 2004; cited by SKM, 2010b). For river red gums, a critical depth of 4 m may also apply (SKM, 2010b).

The germination and establishment of river red gums are not completely dependent on flooding, but regeneration (recruitment success) is greatly enhanced by appropriate flood regimes (i.e. follow-up flooding; Bacon et al., 1993; Roberts and Marston, 2000; MDBC, 2001). Jensen et al. (2008b) reported germination of river red gums following rainfall at an ephemeral creek site on Chowilla Floodplain, indicating that flooding may not be a pre-requisite for germination. However, George (2004, cited by Jensen et al., 2008) suggested that a flood followed by above-average rainfall is a pre-requisite for successful recruitment. Reduction in seedling loss (death) through removal of pressures such as grazing by animals may also be necessary to ensure the viability of the tree community. These floodplain eucalypts retain most of their seed in the canopy, and trees in poor condition produce less seed than those in good condition (George et al., 2005; Jensen et al., 2008a). Where there is widespread loss of mature trees, there is a dramatic reduction in the seed bank to support regeneration.

4.7.2. Influence of regulation

A decline in the condition of native overstorey vegetation along the lower Murray has been recognised for decades (Jolly et al., 1993). In 1990, before the onset of the Millennium Drought, Margules and Partners et al. (1990) estimated that 18,000 ha of floodplain vegetation was severely degraded. Subsequent surveys prior to 2010–11 estimated that over 70% of riparian tree vegetation in wetlands and floodplains in the Murray–Darling Basin are in poor health or dead (Smith and Kenny, 2005; Cunningham et al., 2009). Assessments of tree condition (river red gum, black box and river cooba: *Acacia stenophylla*) at Pike Floodplain in 2009 (Wallace, 2009) revealed that although there were individual trees in 'good' condition, there were no transects (n = 30 trees per transect) with mean or median scores in this category. The most frequently occurring category was 'stressed', with nine of 21 transects in this category.

Drivers for the decline in condition include the construction of instream structures (dams and weirs), abstraction for irrigation and domestic purposes and supra-seasonal rainfall deficits. The combination of these factors has caused a reduction in flooding frequency and an increase in soil
salinisation (Overton et al., 2006; Cunningham et al., 2007; Mac Nally et al., 2011). Shallow depth to saline groundwater is a more substantial stressor in the lower Murray than the upper and mid-reaches. The lower Murray floodplain is underlain by highly saline groundwater, and the depth to groundwater has been reduced as a result of elevated surface water levels in the main channel caused by weirs and by irrigation. Evapotranspiration has caused salt to accumulate in the upper layers of the soil. Under natural conditions, flooding would leach the soil, but reduced flooding frequency and below average rainfall have prevented this. The Millennium Drought increased the stresses on trees that were already in decline (Jolly et al., 1993; Doble et al., 2006; Overton et al., 2006; Holland et al., 2009).

The age-class distribution of woodland trees reflects recruitment and survival, and the growth of young trees must at least match the mortality of older trees if a stand is to remain viable (George et al., 2005). Assessments of population demographics at wetlands in the lower Murray (Aldridge et al., 2012b), Banrock Station (George et al., 2005), Pike Floodplain (Wallace, 2009) and Chowilla Floodplain (Wallace, 2013b) confirm that there is insufficient recruitment to sustain the existing forest and woodland communities.

4.7.3. Influence of management

Tree condition

The capacity of trees to respond to EWAs is coupled to pre-existing condition. As part of an assessment of the efficacy of EWAs specifically for improving the condition of river red gums, Wallace (unpublished) demonstrated that 92% of trees with 1–50% canopy cover responded to the application of environmental water, and 68% of trees that had recently lost all of their canopy (i.e. within several weeks) also responded. However, <15% of trees that had lost their canopy and were showing signs of inner bark cracking responded. At the site scale (independent of individual tree condition prior to the EWA), 80% of trees had responded by 8 weeks postwatering, but by 32 weeks post-watering only 70% of trees were still showing positive signs. In this case, ensuring appropriate application of water while trees retain >50% of their canopy may represent a threshold for intervention. Delivery of water at this time may retain resilience such that a follow-up flood is not required for 2–3 years. However, once trees have lost most of their canopy, higher frequency flooding may be required to ensure recovery. Repeated watering is required to maintain the initial response and to ensure that tree health continues to improve (Overton and Doody, 2008; Souter et al., 2013).

Creation and maintenance of freshwater lens

Freshwater lenses are formed and maintained by a combination of lateral (horizontal) and vertical infiltration of flooding, and to a lesser extent by rainfall. Existing conceptual models of groundwater recharge during floods suggest that inundation of large floodplain areas will recharge a shallow aquifer once a hydraulic gradient is established between the floodwaters and the aquifer (SKM, 2011). Vertical recharge occurs through direct infiltration of floodwater, and horizontal recharge occurs in wetlands, backwaters, the river and its anabranches and tributaries when the surface water level exceeds the head at the groundwater interface (Holland et al., 2006). Recharge decreases groundwater salinity, as the hydraulic gradient is reversed and fresh water flows into the more saline floodplain aquifer (Hantush, 2005; Jolly et al., 1998; Woessner, 2000). These effects extend up to 50 m from the stream edge (Holland et al., 2006).

The magnitude of the lenses is dependent on flood frequency and duration, primarily due to changes in the rate at which water infiltrates through contrasting soil types. On the floodplain, it is likely that these lenses decrease *via* evaporation and evapotranspiration in the absence of flooding, leading to plant stress (Telfer et al., 2008). More frequent inundation and the associated vertical and lateral infiltration of fresh water will flush the unsaturated and saturated zones, providing a freshwater lens to improve soil moisture availability for the vegetation.

4.7.4. Flow-inundation curves

Flow inundation curves (Figure 4.10) show that the area of black box inundated has a very slow linear response rate (a = 0.0940) for flows of 20,000–40,000 ML day⁻¹, with area inundated increasing from 154 ha (0.98% of total area) to 241 ha (1.5% of total area). In contrast, the curve for river red gum is exponential for flows of 20,000–40,000 ML day⁻¹, and the area inundated increases from 1580.7 ha (8.36% of total area) to 2614.2 ha (13.82% of total area).

A summary of the difference in availability of suitable habitat for submergent aquatic vegetation, river red gum and flood-dependent understorey vegetation, for the lower (3000 ML day⁻¹) and upper (40,000 ML day⁻¹) extent of in-channel flows is shown in Figure 4.11. The spatial distribution assumes:

- Submerged vegetation: Photic depth 1 m. In low flows, distribution is up to 1 m below pool level. At higher flows (e.g. 40,000 ML day⁻¹), plants are limited to the lower pool; in the mid- and upper-reaches, plants are extirpated by levels ≥1 m above pool level.
- River red gums: The area where the condition of adult river red gums is maintained or improved is ≤1m above pool level. Note that recruitment can potentially occur between pool level and just below the maximum water level.
- Flood-dependent understorey: At low flows, plants will occur in a narrow band just above the water (where there is no *Phragmites* or *Typha*). At higher flows, they will be present after water levels fall in the inundated area.

4.8. Temporary wetlands

4.8.1. Background

Floodplain wetlands include habitats from permanent and semi-permanent channels to the ephemeral floodplain (Amoros and Bornette, 2002; Ralph and Rogers, 2011), characterised by high temporal variability, driven by wetting and drying cycles of expansion and contraction of habitat (Kingsford, 2000; Gawne and Scholz, 2006). The variations in permanence, depth, water quality, riparian vegetation, habitat complexity and lateral connectivity interact (Boulton and Brock, 1999), so that wetlands are sites of high biodiversity (Arthington et al., 2010). The wetting and drying cycles determine the available habitats. During the wet phase, ephemeral wetlands provide breeding habitat for a range of biota. During dry phases, wetland basins and fringe zones provide critical habitat for terrestrial birds, mammals and reptiles, and function as seed banks for plants and aquatic invertebrates.

Vegetation

Wetland vegetation provides habitat and contributes to primary and secondary production, soil stability, nutrient cycling and water quality (Baldwin et al., 2005). As the distribution and

abundance of wetland vegetation are strongly influenced by hydrology, alterations to wetting and drying cycles are likely to impact on these assemblages and ecosystem functions (e.g. nutrient cycling, soil condition, provision of food resources, structural habitat). Stabilisation of water levels has constricted the bands at which specific water regimes occur, so that plants with particular water requirements are constrained to narrow bands where those requirements are met. Vertical and lateral infiltration of surface water maintains freshwater lenses and increases soil moisture availability to support fringing woodlands (section 10).



Figure 4.10. Flow-area inundation plots for floodplain eucalypts. Note the different axes for A and B. For flows of 20,000–40,000 ML day-1 (B), there is a linear relationship for black box: y = -3338 + 0.094x, and an exponential relationship for river red gum: $y = 1486.6 + 7.7391 e^{0.0001x}$.



Figure 4.11. Availability of suitable habitat for submergent aquatic vegetation, river red gum and flood-dependent understorey vegetation, relative to surface water level for [A] 3000 ML day⁻¹ and [B] 40,000 ML day⁻¹. See text for assumptions regarding distribution.

Hypoxia and anoxia

Hypoxia and anoxia (low and zero dissolved oxygen concentrations, respectively) are a major concern for the ecology of wetlands, as tolerance to low oxygen is species and life-stage specific, and changes in DO can affect biodiversity (Ekau et al., 2010). Hypoxia is associated with fish kills (Erskine et al., 2005), disruption of endocrine systems (Wu et al., 2003), embryonic development (Shang and Wu, 2004) and survival and hatch rates of fish (Hassell et al., 2008) and degradation of lotic macroinvertebrate communities (Feminella et al., 2003; Walsh et al., 2001; Walsh, 2002) and wetlands (Spieles and Mitsch, 2003). Hypoxia is also significant for freshwater mussels and crayfish, and is one of the factors that restrict *Alathyria* (river mussel) and *Euastacus* (Murray

crayfish) to channel rather than floodplain habitats. In contrast, *Velesunio* (floodplain mussel) and *Cherax* (yabbies) are comparatively tolerant to low DO (Walker, 2006).

Anoxia may release sediment-bound material such as manganese, iron (Davison, 1993), ammonium (Lawrence and Breen, 1998; Boulton and Brock, 1999; Morin and Morse, 1999) and phosphorus (Mortimer, 1941; Laws, 1993; Martinova, 1993), convert dissolved organic nitrogen to ammonia and nitrate (Harris, 2001) and release redox-sensitive compounds from sediments (e.g. Baldwin and Mitchell, 2000; Dahm et al., 2003), some of which (e.g. ammonium, sulfide) are toxic to aquatic organisms (Vismann, 1996; Hickey and Martin, 1999).

Larvae and young-of-year fish may survive at DO concentrations as low as 20% saturation (e.g. 1.8 mg L^{-1} at 20°C), but growth is restricted (Gehrke, 1988, 1991) and fish are likely to die at DO concentrations < 1.5 mgL^{-1} (Miranda et al., 2000). Degradation of lotic macroinvertebrate communities occurs under hypoxia (Feminella et al., 2003; Walsh, 2002; Walsh et al., 2001; Walsh, 2002) and wetlands (Spieles and Mitsch, 2003). The influence of hypoxia on wetland microinvertebrates is less widely studied but research (e.g. Mikschi, 1989; Roman et al., 1993; Watkins et al., in review) suggests that community structure would be substantially altered (Davidson et al., 1998).

Food resources

Factors that influence the success (survival and recruitment) or failure of breeding events of groups such as frogs, fish and birds include the availability of appropriate food resources at the correct times *via* the productivity 'boom' that occurs during floods (Bunn et al., 2006). The boom provides abundant food and is therefore an ecosystem 'service'. Invertebrates are food for breeding waterfowl, as they provide protein for egg and nestling development. The responses of bird that are piscivorous, herbivorous, reliant on aquatic macroinvertebrates and terrestrial invertebrate/ insects, or utilise aquatic plants (e.g. sedges and rushes) for nesting material will depend on the provision of appropriate habitat and response/development of food resources (Rogers and Paton, 2008).

Failure to establish a diverse and abundant zooplankton and macroinvertebrate community within floodplain wetlands may lead to a lack of food resources for higher trophic levels that normally would benefit from the boom associated with inundation of floodplain habitats. Hypoxia causes deformities and death of embryos of some frog species, early hatching may have a negative impact on growth, ability to avoid predation and reproductive success at adult stages (Mills and Barnhart, 1999; Seymour et al., 2000).

Fish

During the inundation phase, large, deep wetlands may provide fish recruitment zones that are rich in food resources and structural habitat. During the drying phase, declining water quality and spatial extent result in lower growth rates and/or increased predation. Although the role of floodplains for native fish remains in some contention (section 9), Beesley et al. (2011) reported high concentrations of nutrients, chlorophyll *a* and microcrustaceans in semi-permanent wetlands compared to the channel in the mid-Murray. High food density coupled with low predation and competition should increase larval survival and growth. The authors concluded that long filling periods (weeks–months) are associated with increased recruitment and body condition of small-bodied fish (section 9).

Many wetland specialist species with conservation listing (e.g. Murray hardyhead, purplespotted gudgeon, southern pygmy perch, Yarra pygmy perch) prefer wetlands and their declines may be due to lack of intermittent wetland habitat, the loss of lotic riverine conditions or both (Lintermans, 2007).

Waterbirds

Birds often are at the top of food webs, and are able to fly in/out in response to changing conditions. Consequently, their responses to environmental change tend to be indirect, manifest through the responses of trophic levels below them. Benefits to bird communities from the delivery of water into temporary wetlands depend on the organisms on which the food web relies, such as aquatic and terrestrial vegetation, macroinvertebrates and fish (Rogers and Paton, 2008). For example, waterbirds dependent on invertebrates will respond to flooding prior to those dependent on macro-crustaceans or macrophytes (Kingsford et al., 2010). Observations of breeding in artificially watered wetlands throughout the Murray-Darling Basin suggest that the lack of natural cues may not be a critical issue for waterbirds (Rogers and Paton, 2008).

The extent of wetland inundation and the size of a waterbird breeding event are linked (Crome, 1988; Briggs et al., 1997; Scott, 1997, cited by Rogers and Paton, 2008), as large areas of productive wetland are able to support more nests (Rogers and Paton, 2008). Water depth has a significant influence on waterbird diversity, as particular species occupy specific habitat types defined by depth. Deep habitats benefit diving piscivorous birds. Shallow (<50 cm) wetlands offer a habitat for feeding compared to the main channel which does not support the diversity of water regimes or extent of appropriate habitats (Brookes et al., 2009). Wetland management that provides the greatest diversity of habitats including variable water depths, mud flats, inundated vegetation and deeper water areas will promote the greatest abundance and diversity of waterbirds (Brandis et al., 2009).

Waterbirds can be classified into six groups, based on feeding and habitat requirements (adapted from Waanders and Kuchel, 2011)).

- *Waterfowl*: reliant on submerged vegetation and aquatic invertebrates (e.g. swans, ducks, geese, gallinules, coots)
- *Waders and shorebirds*: forage in shallow water (c. 0.3 m) or over-exposed shorelines (e.g. ibis, spoonbills, egrets, sandpipers)
- *Piscivorous birds*: hunt for fish while flying over or sitting on a water body (e.g. grebes, darters, cormorants, pelicans, egrets, bitterns, herons)
- *Cryptic birds*: reliant on dense riparian vegetation for cover (e.g. rails, Australasian bittern)
- *Gulls and terns*: hunt while flying and dip to the water surface to catch prey (e.g. silver gulls, gull-billed and white-winged terns)
- *Reed-dwelling passerines*: not technically waterbirds, but use riparian reeds beds for habitat (e.g. reed-warblers, grassbirds).

The duration of flooding is the primary determinant of the success of a nesting event. Minimum inundation times for waterbirds typically are around 100 days, but for some species (i.e. black swan) they may be as high as 180 days. The time required varies between species based on intrinsic (physiological preparation, egg development and incubation, nestling development) and extrinsic (development of food/prey communities) factors that are influenced by spatio-

temporal variability in wetland productivity (Rogers and Paton, 2008). A minimum of 120 days (4 months) is suggested as a minimum (Young et al., 2003, cited by Rogers and Paton, 2008).

Frogs

Eleven species of frogs occur in the Murray corridor (Cale, 2008; Veltheim et al., 2009):

- Eastern sign-bearing froglet (Murray Valley froglet) Crinia parinsignifera,
- Common froglet Crinia signifera,
- Eastern banjo frog Limnodynastes dumerilii,
- Long-thumbed Frog Limnodynastes fletcheri,
- Spotted grass frog Limnodynastes tasmaniensis,
- Peron's tree frog *Litoria peronei*,
- Southern brown Tree Frog Litoria ewingii,
- Southern bell frog Litoria raniformis,
- Painted frog Neobatrachus pictus,
- Sudell's frog *Neobatrachus sudelli* and
- Brown toadlet (Bibron's toadlet) Psuedophryne bibronii.

Anecdotal evidence indicates declines in the abundance of frogs in the Riverland, but there is no evidence of local extinctions (Herbert, 2000, as cited by Cale, 2008). The golden bell frog (southern bell frog) *Litoria raniformis* is listed as vulnerable under the *Environment Protection and Biodiversity Conservation Act 1999* and the SA *National Parks and Wildlife Act 1972*.

Floodplains provide key aquatic and terrestrial habitats for frogs. Amphibians in semi-arid environments breed readily in ephemeral waterbodies which can be highly productive systems with low predator densities (Babbit and Tanner, 2000). For many species, inundation of floodplain habitats is essential as they depend on surface waters at a suitable time of year to allow tadpole development and metamorphosis. Heavy rainfall may support breeding of some species (e.g. *Neobatrachus* spp.) but not wetland species.

Six variables are important for frogs, namely season, duration, extent, frequency, water quality and predation. Changes in the timing and extent of flooding on the lower Murray floodplain are likely to have influenced breeding success and survival (Cale, 2008). Existing populations can be affected by circumstances which reduce the magnitude, timing and duration of inundation of floodplain wetlands, and hence reduce the available breeding habitat. Conversely, the provision of environmental water to these areas can trigger breeding and lead to population recoveries.

Frogs and tadpoles are strongly associated with wetland vegetation, as it provides food, sites for deposition of eggs for some species and cover/protection from predators. Most frog species that occur in the southern Murray-Darling Basin have wide breeding seasons, but few breed during June and July and water delivered during this period is likely to have less effect than water delivered in August–September. For frogs, water needs to persist in wetlands for a minimum of 3–5 months (90–150 days) to allow sufficient time for metamorphosis (Veltheim et al., 2009). For some species, such as *Crinia* spp., shorter periods (6 weeks) may be sufficient for metamorphosis (Anstis, 2007). Providing quality habitat at multiple sites reduces the adverse effects of pressures such as disease, poor-quality habitat or predation.

Frog species richness and recruitment in the lower Murrumbidgee River differ among habitats, depending on watering frequency. Habitats subject to intermediate levels of flooding support

more species. This suggests that high primary production and low predation are drivers (Wassens et al., 2007). Eastern gambusia (*Gambusia holbrooki*) is a small bodied, invasive fish that tolerates a wide range of temperature, oxygen, salinity and turbidity. It is aggressive and will fin-nip larger fish and prey on the eggs of other frogs and native fish. It is implicated in the decline of more than 10 species of frogs in Australia (Lintermans, 2007).

4.8.2. Influence of regulation

Unregulated river systems have a high degree of resilience, latitude and resistance, displaying a transient, dynamic regime (Holling, 1973) with two extremes (Scheffer et al., 2001; Scheffer and Carpenter, 2003) in which wetlands are always drying or flooding (Kingsford et al., 2010). Rather two states with characteristic dominant biota, there is a single state with two alternative phases, interspersed by floods and droughts (Colloff and Baldwin, 2010). Once the disturbance (flooding or drying) is removed, the system progressively reverts to the preceding condition. Once a driver (permanent inundation or very long drying) exerts sufficient pressure, a catastrophic rather than smooth transition to an alternate state can occur (Scheffer et al., 2001; Scheffer and Carpenter, 2003). This concept is illustrated in Figure 4.12. The unregulated system is represented by the middle box (normal dynamic range). Management has skewed river channels towards the left ('anti-drought'). About 75% of wetlands between the SA Border and Wellington are connected to the channel at normal pool heights (Pressey, 1986; Jones and Miles, 2009) and are included in the anti-drought group. Floodplains and ephemeral wetlands with sills above normal weir-pool height are skewed to the right ('engineered drought').



Figure 4.12. Conceptual model of the dynamic regime in which floodplains are always drying or flooding and the potential for excessive pressure to cause a transition to an alternate state. From Wallace et al. (2011).

4.8.3. Influence of management

It is important that individual wetlands are not managed in isolation of the terrestrial floodplain, and that inundation is not provided solely *via* low–moderate flows and connecting channels. Overland flows that connect the channel, wetlands and the broader floodplain are vital. Wide variation in permanence, depth, water quality, riparian vegetation, habitat complexity and lateral connectivity interact such that wetlands are sites of high biodiversity. Periods of high flow that connect the channel, wetlands and woodlands as parts of one ecological system (Lytle and Poff, 2004; Walker, 2009) are drivers of energy flux. Inundation of temporary wetlands creates more productive conditions than those in permanent wetlands (Junk et al., 1989) with the exchange between the river and the riparian zone being a key component of ecosystem function (Vannote et al., 1980). During the rising limb of the hydrograph, material is transported from the river to the wetland. In the recession phase, carbon, nutrients, plankton, propagules and fish are transported from the wetland to the river (Junk et al. 1989).

Delivery of water to ephemeral wetlands also needs to take account of soil salinity (section 10) and the potential impact of inundation on soil sodicity. Sodic soils contain a high proportion of sodium cations relative to other cations (e.g. magnesium, calcium, potassium), and form when magnesium and calcium cations are displaced from clay particles by sodium cations. The displaced cations are leached from the soil profile and the sodium cations accumulate in the soil. Sodic soils contain sufficient sodium cations to affect soil structural stability, reduce infiltration and soil moisture holding capacity and increase soil strength. High soil strength restricts root elongation and survival of plants. Sodicity can substantially reduce infiltration and lead to an increase in salts and elements toxic to plants. It is important to recognise that sodicity is a separate issue from salinity. A soil may be sodic without being saline, saline without being sodic, or both sodic and saline (Rengasamy et al., 2009). High soil salinity and sodicity leads to a loss of soil structure, microbial diversity and abundance (Fitzpatrick et al., 1994), loss of soil egg banks of flood-responsive understorey vegetation and microinvertebrates (Brock et al., 2005b; Nielsen et al., 2003a; Porter et al., 2007).

4.8.4. Flow-inundation curves

The flow inundation curves (Figure 4.13) show two step-changes in the area of ephemeral wetlands inundated, 25,000–30,000 and 45,000–50,000 ML day⁻¹. The first change increases the area flooded from 2381 ha (37.3% total area) to 2791 ha (43.8% of total area).



Figure 4.13. Flow-area inundation plots for ephemeral wetlands. Note the step-change in area inundated for flows of $25,000-30,000 \text{ ML day}^{-1}$.

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