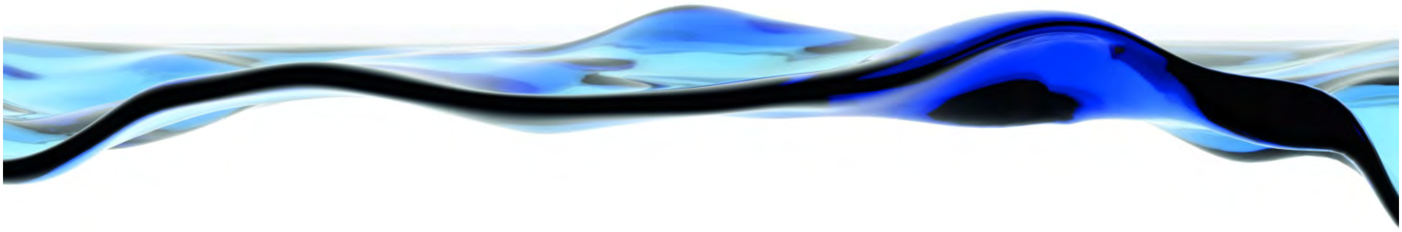
Preliminary systems inventory and project scoping
River Murray Catchment



Goyder Institute for Water Research
Technical Report Series No. 13/9

GOYDER INSTITUTE FOR WATER RESEARCH

www.goyderinstitute.org

Goyder Institute for Water Research Technical Report Series ISSN: 1839–2725

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Citation

Kirby M, Bice C, Doody TM, Hemming S, Holland KL, Jolly ID, Mason K, McGinness H, Muller KL, Nicol JM, Pollino CA, Rigney D, Wallace TA, Ye Q. 2013. *Preliminary Systems Inventory and Project Scoping River Murray Catchment,* Goyder Institute for Water Research Technical Report Series No. 13/9

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Contents

1. Introduction

We report here the findings of the Goyder Institute Project E.1.5 " Preliminary systems inventory and project scoping River Murray Catchment".

The project was commissioned by the Goyder Institute as a key part of its River Murray Programme[1]. The purpose of this project was to scope out potential follow–on project(s) by providing an inventory of knowledge and knowledge gaps, on:

- flows within the Murray–Darling Basin (MDB), the resulting flows into South Australia (SA), and how they are affected by climate change, SDLs, environmental flow management and other influences;
- constraints and opportunities presented by those flows for environmental and Indigenous flow management within South Australia; and
- relationships between the knowledge and knowledge gaps and SA departmental priorities for environmental and Indigenous flow management.

Researchers in the project team each contributed expert knowledge of different aspects of environmental flows in the lower River Murray in South Australia. Knowledge areas included in this report were MDB hydrology, river flows, salinity and modelling (Mac Kirby, Ian Jolly, Kate Holland), Murray–Darling Basin Plan (Carmel Pollino), floodplain overstorey vegetation (Tanya Doody), aquatic and floodplain understorey (Jason Nicol), birds and mammals (Heather McGinness), frogs (Kerri Muller and Kate Mason), fish (Qifeng Ye and Chris Bice), micro- and macro-invertebrates (Todd Wallace), carbon and nutrients (Todd Wallace), algal blooms (Todd Wallace), biofilms (Todd Wallace), drought and salinity (Todd Wallace), indigenous flows (Steve Hemming and Daryle Rigney), predictive ecology (Carmel Pollino) and options for developing an environmental flow tool (Todd Wallace).

Potentially, several projects could be recommended as suitable for a follow-on project; the Goyder Institute itself suggested ten potential projects when it commissioned the project. A key step in the project was to identify which was the highest priority (and why). This was done at a stakeholder workshop in early April 2012; the stakeholders included SA government departmental staff, catchment authority representatives and representatives from indigenous groups, as well as the research organizations of the project team, and the Goyder Institute. The stakeholder workshop, including participants and recommendations, is described in Appendix 1.

The first workshop resulted in a reasonable consensus for a priority project on floodplain processes and their interaction with the river and terrestrial environment; importantly, the focus should be at the scale of the whole SA River Murray floodplain, and the connectivity along the river. It was also agreed that a Phase 1 project should concentrate on developing a framework of understanding; a Phase 2 project should be considered to develop a practical tool for implementation in SA departments. It was emphasized in the workshop that there is a reasonable understanding of, and reasonable management procedures for, individual environmental sites: it is the whole riverine landscape perspective and its connections that are important.

Subsequent workshops were help with key SA government departmental staff in October, November and December 2012 to discuss the proposed follow-on project in further detail. It was agreed that the spatial scale of interest was the floodplain and river channel from the SA border to Wellington, with the

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[1] Goyder Institute Annual Research and Development Plan and Budget 2011/12. Available at http://goyderinstitute.org/uploads/Annual%20R+D%20Plan%20and%20Budget%202011_12%20v4_4.pdf

importance of developing future linkages to the Coorong, Lower Lakes and Murray Mouth ecosystem. State government staff stated that the project needed to develop a tool or model that works at an operational time scale in order address questions about ‘How best to deliver an event moving through the system? What are the priorities of delivery for that water?’ It was also emphasized that the project needs to identify knowledge gaps, seeking to fill these where possible, and recommending monitoring that could test assumptions and fill knowledge gaps. The issue of salinity and its interactions with ecology was also raised, including ‘What are the sources of salinity? Where is salt likely to be flushed? How does this relate to ecological responses?’ It was decided that there was a need for a spatially explicit salt mobilisation risk tool, however this was outside of the scope of this follow-on project and should be considered with other risks, including blackwater and algal blooms in the future. It was proposed that the follow-on project would develop an integrated model of hydrology, water quality and ecology to assess the ecological benefits of watering scenarios at a landscape scale and at a temporal scale relevant to operational decision making.

Based on the outcomes of these workshops, in this report we first assess in Chapter 2 the flows in the Murray–Darling river system, the likely future changes to flows and discuss what that means for flows into South Australia and the management of those flows. In Chapter 3 we describe the relevant literature and the knowledge gaps on riverine and floodplain ecology in South Australia. We have focussed on the flood pulse response as this is the key driver of change in the floodplain – river ecological communities of the lower River Murray in South Australia. In Chapter 4 we discuss requirements for identifying and specifying indigenous flows and environmental flows. Then, in Chapter 5 we discuss predictive ecology – going from knowledge of the underlying ecological concepts to models that allow us to link flows to ecological impacts – and assess the state of knowledge and models for South Australia. In Chapter 6, we describe options for developing an environmental flow tool based on ecological objectives and targets. The report concludes with a description of knowledge gaps identified in this report and in recent Murray–Darling Basin Authority commissioned literature reviews on environmental watering.

2. The Murray–Darling Basin

2.1. River flows

The hydrology of the Murray–Darling Basin, and the impact of changes such as climate change, the drought and sustainable diversion limits, has been described in detail in many recent studies including CSIRO (2008), CSIRO (2011), and Kirby *et al.* (2012a, 2012b). Here we summarise the main issues relevant to flows and possible future changes to flows into South Australia.

The Murray–Darling Basin occupies just more than one million square kilometres and, in an average year, receives just less than 500 mm of rainfall; the total volume of rainfall is thus about 500,000 GL. Most of the rainfall evaporates or is transpired by plants before it becomes runoff. The rainfall is greatest and the evapotranspiration least in the south–eastern highlands, and this area is the source for the greatest amount of runoff (Figure 2.1).

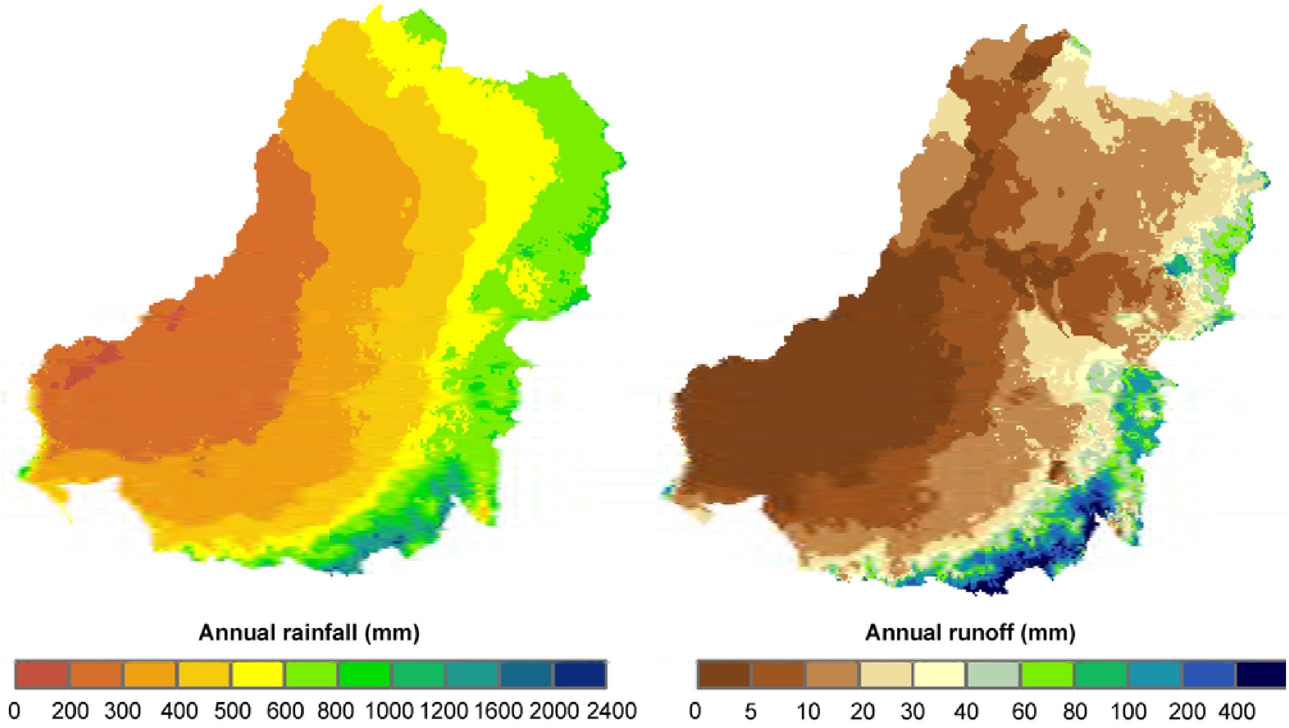

Figure 2.1 Distribution of averaged annual rainfall and runoff in the Murray–Darling Basin. Source: CSIRO, 2008.

As well as being generally low, rainfall is highly variable, with frequent long droughts and wetter periods with significant floods. One consequence is that the two main rivers, the Murray and the Darling, have highly variable flows (Figure 2.2), with the Darling being one of the world's most variable rivers (Puckridge *et al.* 1998). Figure 2.3 shows the daily flow rate into South Australia, which results from the combined flows of the Murray and Darling, though most of the flow is contributed by the Murray. The impact of the recent drought can be seen in Figure 2.3, with almost no sizable flow between 2000 and 2010.

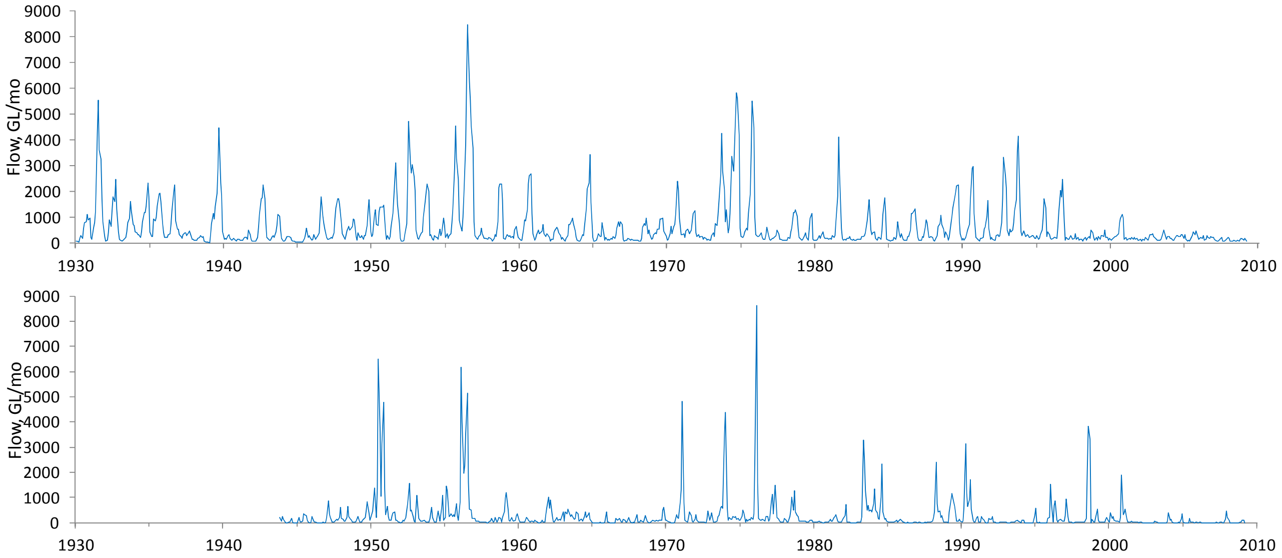

Figure 2.2 Measured monthly flows (in GL mo^{-1}) for the River Murray at Euston (top) and the Darling River at Bourke (bottom). The Darling measurements start in 1943. Source: NSW Water Information, http://waterinfo.nsw.gov.au/ (under Real Time Data), and Victorian Water Resources Data Warehouse, http://www.vicwaterdata.net/vicwaterdata/home.aspx.

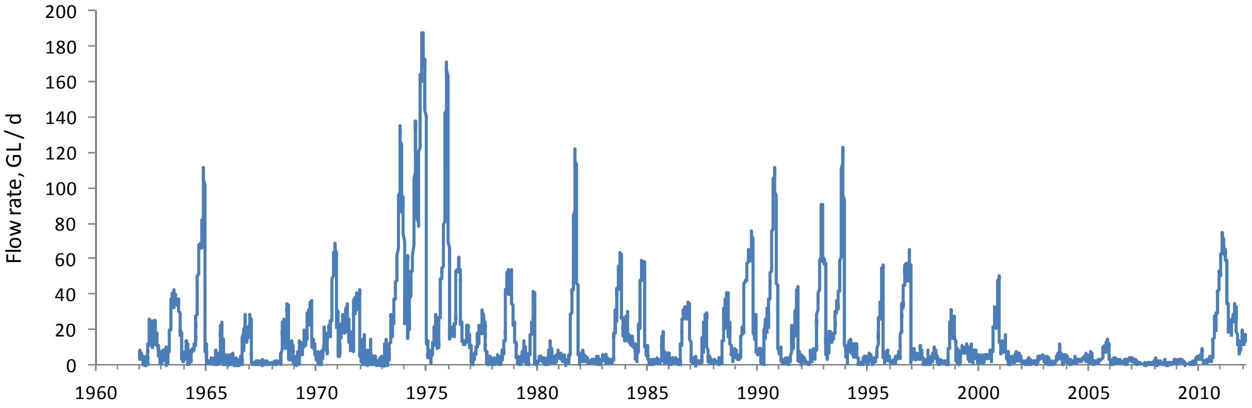

Figure 2.3 Daily flow rate at Lock 9. Source: Department for Water, WaterConnect, https://www.waterconnect.sa.gov.au/RMWD/Pages/default.aspx.

An approximate water balance based primarily on measured flows in the main river channels of the Murray–Darling Basin is given in Table 2.1. Modelled rainfall-runoff estimates were also used to estimate inflows. Discharge at the mouth is not measured, but is estimated to be about 1,600 GL yr^{-1} less than that measured at Lock 1. The lack of balance in the account is due to measurement errors. Other water balances based on estimated runoff, such as the National Water Account 2010 (BoM, 2011) give larger estimates of water availability, and correspondingly larger estimates of river and floodplain losses; the difference arises because some of the runoff is already lost by the time it reaches the inflow points used to construct the water balance in the table. Table 2.1 shows that diversions are a large part of water use, accounting for over 40% of total inflows. Diversions include water used for human consumption, irrigation and stock and domestic water supply. River and floodplain losses include water that is lost from side channels and from the main river channel as lateral bank recharge, overbank spills (floods) that do not return to the river and are consumed as evapotranspiration on the floodplain, and vertical seepage to groundwater. Reach losses are water lost as evaporation directly from open water in the main river channel. River and floodplain losses and discharge all reduced substantially in the drought.

Table 2.1 Approximate measured annual average water balance in GL yr^{-1} for 1990–2006 for the main river channels of the Murray–Darling Basin. Source: Kirby *et al.* (2008).

Inflows	**23,403**
Discharge at Lock 1	6,156
Diversions	9,393
River and floodplain losses	3,667
Reach evaporation	3,102
Total outflows	**22,317**

The impact of the Millennium drought

During the drought, inflows into storage dams, flows in the river, and irrigation diversions all reduced. Figure 2.2 shows the much reduced flows after 2000 in the Murray and Darling rivers. The impact on irrigation diversions and modelled flow at the barrages, which is the equivalent of discharge from the Murray Mouth is seen in Figure 2.4. With the onset of major declines in water availability from 2002, flow at the barrages or discharge from the mouth declined almost to zero, whereas irrigation diversions declined somewhat to 2006, and then dramatically following the record reduction in inflows in the later part of the drought.

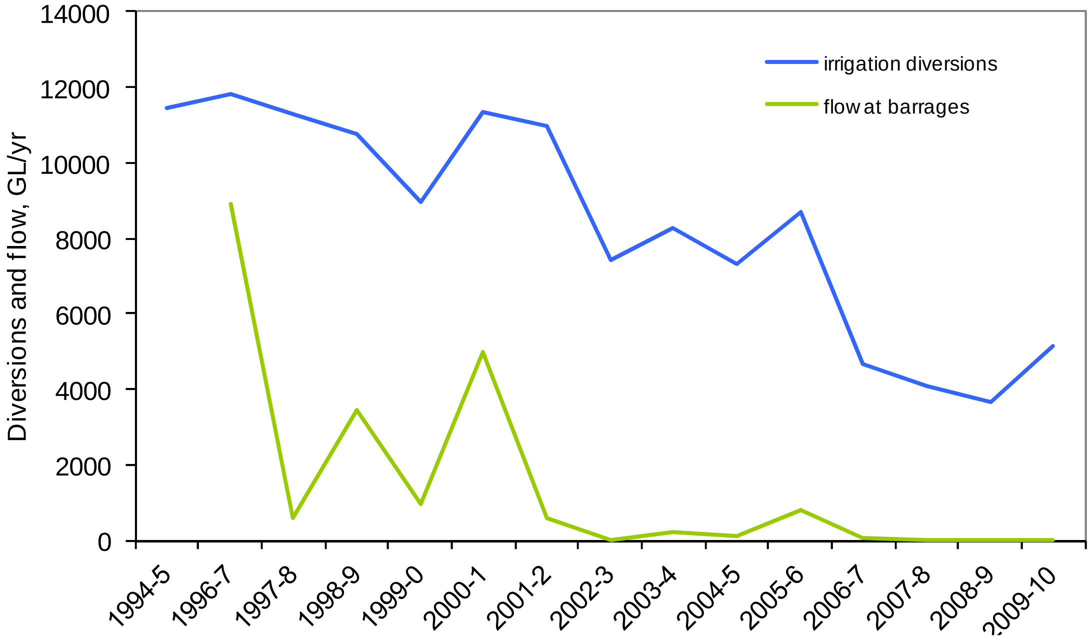

Figure 2.4 Diversions from the Murray–Darling Basin and modelled flow at the barrages or discharge from the Murray Mouth, 1994–5 to 2009–10. Source: MDBA, 2011a.

The impact of water trading

Most water trading, whether of entitlements or allocation, is local, and has little impact on flows into South Australia (MDBA, 2011a). Long distance trading is mostly of allocations, and mostly sales from the middle reaches of the Murray, Murrumbidgee and Goulburn, to buyers in the lower reaches of Victoria and South Australia. Figure 2.5 shows that the net trade into South Australia was minor before 2006/07, rising to about 330 GL in 2008/09, and declining to 235 GL in 2009/10. The economic impact of the purchased water was to largely maintain irrigation activity and the gross value of irrigation in South Australia at pre–drought levels, despite the lower allocations of water (Kirby *et al.*, 2012a). It is not yet known whether the extent of downstream trading was a one–off response to the drought, or whether it is a new and permanent behaviour now that trading is commonplace.

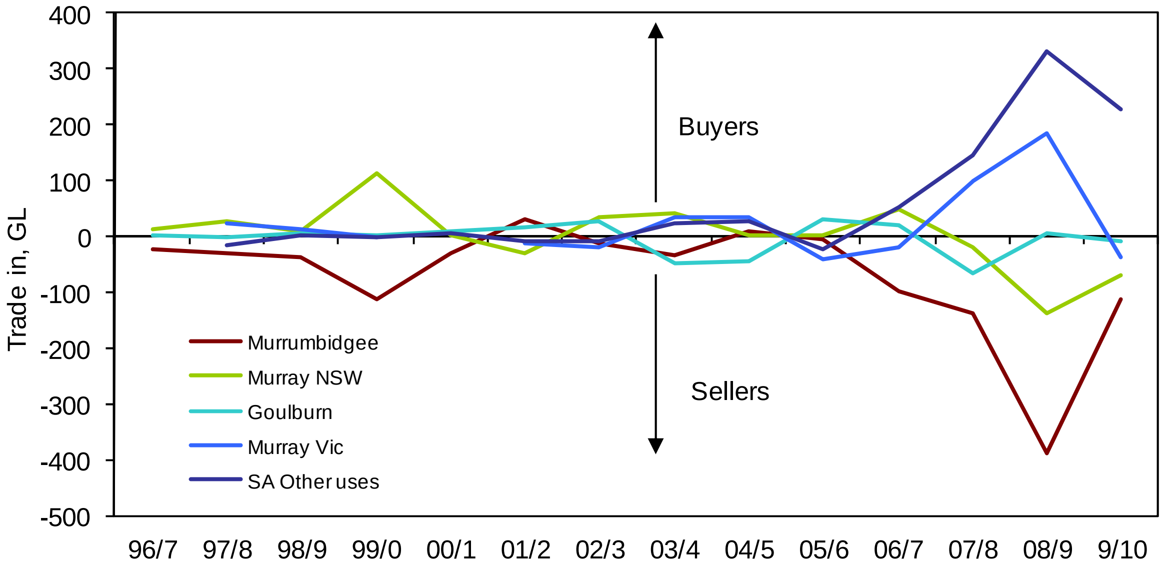

Figure 2.5 Net trade into and out of the main regions in the southern Murray–Darling Basin. Source: MDBA, 2011.

The projected impact of climate change

CSIRO (2008) studied the impact of climate change, including changes to temperature and rainfall. They showed that, while there is considerable uncertainty, most models of projected climate change lead to decreased rain, and hence decreased runoff and river flow, in the southern Murray-Darling Basin by 2030. The wettest scenario reported by CSIRO (2008) was for river flows similar to those in the historic record, while a median scenario was for decreases in river flows of 11 %, and a dry scenario was for decreases in river flows of 34%. Discharge at the Murray Mouth is projected to be affected more than average flows in the river: the projected decline in discharge at the mouth in the median climate change scenario is nearly 25%, compared to a general reduction in river flows of 11%. Kirby *et al.* (2013) considered the CSIRO (2008) projections for future droughts in detail, and concluded that particularly in the south of the basin, future droughts are projected to be more severe and more frequent than those experienced to date: that is, future droughts could reduce flows in the river to a greater degree than the Millennium Drought.

2.2. River salinity

High salinity levels are a long–recognised issue in the lower reaches of the River Murray (Jolly *et al.*, 2001), and in part are a natural consequence of Australia's dry climate and highly weathered landscape that leads to large storages of salt in the soils and groundwater (Figure 2.6). However, clearing of native vegetation and dryland and irrigated agriculture have changed the water balance of catchments. This is due to large increases in recharge to the regional groundwater beneath areas, when deep drainage was estimated to have increased from less than 0.1 mm below native vegetation to ~3 mm below shallow rooted crops and pastures (Allison and Hughes, 1983). Increased recharge has lead to increased mobilisation of salt from the naturally saline regional aquifers into the River valley.

In response, the Murray–Darling Basin Authority (MDBA) (and its predecessor the Murray–Darling Basin Commission (MDBC)) has implemented a number of initiatives to manage the salinity of the river. The Basin Salinity Management Strategy (BSMS) commenced in 2001 with the aim of guiding Basin communities and governments in working together to control salinity and protect important environmental values and assets

(MDBMC, 2001). It built upon the 1988 Salinity and Drainage Strategy (MDBMC, 1989) which, through the implementation of salt interception schemes and the effectiveness of state salinity action and land and water management plans, reduced salinity in the lower River Murray. Figure 2.7 shows a time series of recorded river salinity at Morgan, South Australia along with a modelled salinity time series representing the continuation of 1975 salinity management practices. Morgan is located ~315 river km upstream of the mouth and is just downstream from where the river turns from flowing westward to southwards towards the mouth. Observed salinity levels are almost universally below what they would have otherwise been in the absence of the intervention activities.

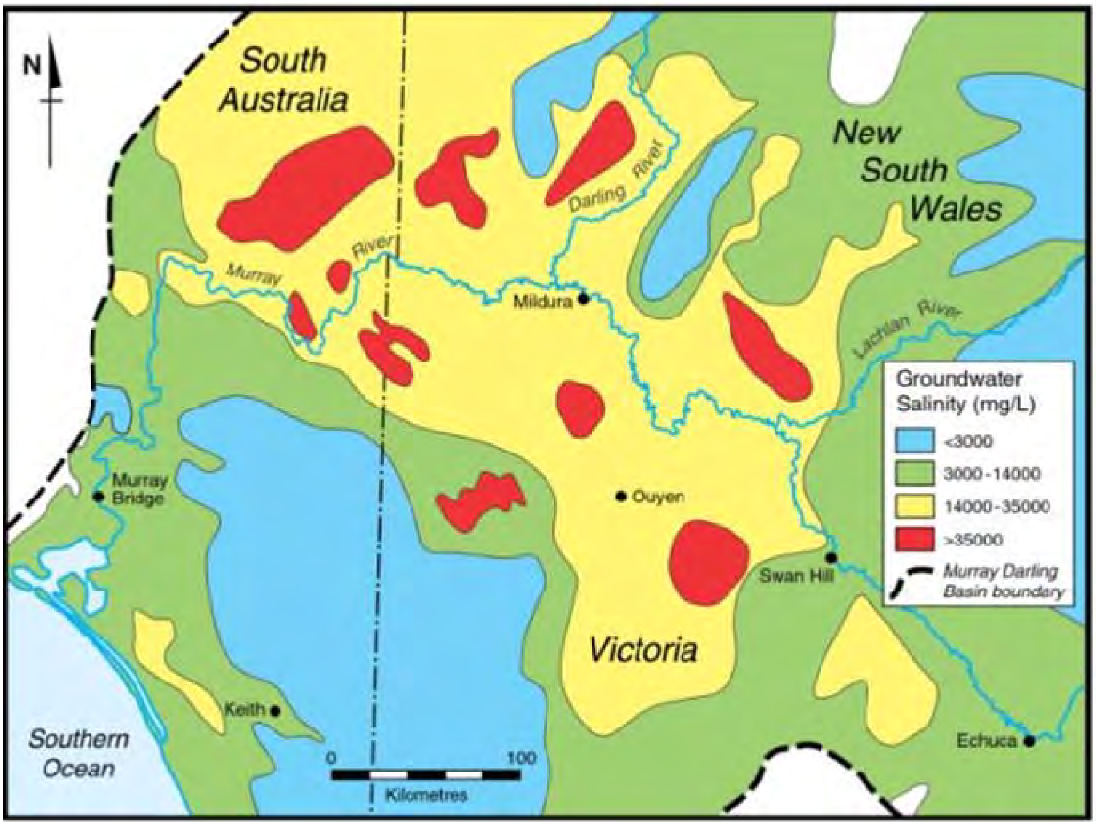

Figure 2.6 Murray-Darling Basin groundwater salinity. Adapted from Evans and Kellett (1989).

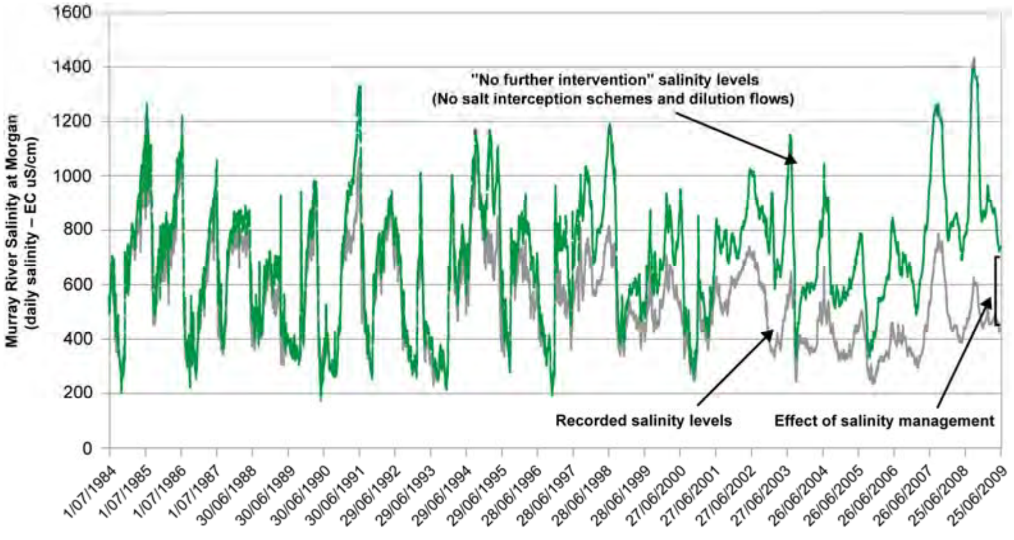

Figure 2.7 The effect of salinity management in the Murray–Darling Basin: comparing recorded mean daily salinity levels at Morgan over a 25 year period (July 1985 to June 2010) with modelled salinity levels

without salt interception schemes, land and water management plan actions and additional dilution flows – the “No further intervention” scenario. Source: MDBA (2011b).

Whilst the implementation of salt interception schemes has stemmed the rise in the salinity of the River Murray in South Australia, there is likely to be significant future increases in salinity as a result of the delayed impacts of clearing in the Mallee region (Cook *et al.*, 2001). Current salt interception schemes are not configured to mitigate these potentially large future increases (up to 50 µS cm^{-1} increase at Morgan by 2100; Barnett and Yan, 2006). Nor are they configured to mitigate the potential River salinity impacts of increased environmental water management activities (refer Section 2.4 below).

2.3. River modelling

Regulated rivers such as the River Murray can be modelled from images of individual floods and integrated into a geographical information system (GIS). The River Murray Floodplain Inundation Model (RiM–FIM) was developed to support environmental flow management in the River Murray. RiM–FIM encompasses the entire length of the regulated section of the River Murray (~2,155 km) from Hume Dam at Albury (Victoria) to Lake Alexandrina at Wellington (South Australia). Inputs to the model are the river flow at each of 22 gauging stations and represent flow data reported from the MDBC MSM–BigMOD flow model. It is useful for predicting the extent of inundation on the River Murray floodplain (~606,000 ha) including the flow regimes of wetlands and floodplain vegetation. The RiM–FIM data includes the floodplain boundary and extent of floodplain inundation. The floodplain boundary follows the active floodplain based on floodplain vegetation extent. It excludes agriculture and population areas that have been protected from flood. Seventy–eight satellite images from Landsat TM imagery were used to construct the model. Several bands and image enhancement were used to distinguish water from other features (e.g. shadow). The images were registered to real–world coordinates so that the flooded areas would coincide or overlap in the GIS. Satellite images provided a range of flows and interpolation between the discrete flow intervals provided a more continuous predictive model. In South Australia the model has been extended to link river flow with water height using a series of backwater curves and a river flow model. The South Australian section has also had an interface developed so that the model can be run from within a GIS package to predict the extent of flooding from different river flows, different weir heights and different times of the year. RiM–FIM has been used to inform managers of floodplain areas inundated by weir pool manipulation of environmental flows (e.g. Siebentritt *et al.* 2004) and in the SA Government assessment of the likely ecological consequences of the Proposed Basin Plan (Bloss *et al.* 2012).

More detailed hydraulic modelling using the two-dimensional, numerical, hydraulic model MIKE 21 has also been undertaken for the lower River Murray in South Australia (Macky and Bloss 2012). MIKE 21 is based on a computational grid that covers the area of interest and the model calculates flow, flow velocity, and water level for each cell of the grid. Input information includes the land surface elevation, surface roughness, initial water level, and eddy viscosity for each grid cell. Model coverage was from Lock 1 to upstream of Chowilla, just past the South Australian state border with New South Wales and Victoria over a total river length of 375 km. Model cell size was 15 m, which was resampled from the existing 2 m resolution DEM. The model was used to compute inundation extents, water surface elevation and bathymetric data. Computed inundation extents were manually edited to remove ‘false positives’ where isolated wetlands are shown as full, but in reality would be dry. The water surface elevation and bathymetric data were used to calculate wetland inundation volumes for weir raising scenarios. Model outputs for the weir pool lowering scenarios were restricted to longitudinal water surface profiles due to the lack of bathymetric data below pool level.

As described above, there are clear ecological benefits from improving the inundation regime of River Murray floodplains and wetlands to mitigate the effects of salt stored in those environments. However, it is well known that floodplain and wetland inundation from natural floods leads to mobilisation of salt stored in these environments into the River Murray in South Australia (Jolly, 1996). These salt accessions to the river can occur for several months after the recession of a flood which, when combined with the reduction in the river's dilution flows, leads to a downstream increase in river salinity. There are short–term salt accessions that occur for the first few days or weeks that originate from processes such as salt wash–off from the surface of floodplain soils and the beds of ephemeral wetlands, flushing of salt stored in the water columns and beds of permanent wetlands, and groundwater mixing processes within bank storage. There are also long–term salt recessions that can occur for many months after the flood peak that are due to the inundation causing groundwater recharge beneath floodplains and wetlands that leads to a slow displacement of saline groundwater to the river and its anabranches (Overton *et al.*, 2005).

It is therefore not unreasonable to assume that some of the future environmental watering activities and other environmental management activities such as weir pool lowering and raising may lead to mobilisation of salt into the River Murray system. This has the potential to pose operational risks for downstream water users who could be exposed to elevated river salinities that limit their use of the water. These environmental activities also pose potential salinity accountability risks to the Salinity Registers operated under the MDBA Basin Salinity Management Strategy (BSMS) 2001–2015 (i.e. salinity impacts at the Basin target site of Morgan in South Australia). Concerns about this issue were raised in the 2007–08 MDBA Independent Audit Group – Salinity (IAG) report (MDBA, 2009). River salinity management is further complicated by increases in river flow for environmental purposes that could potentially dilute salinity levels in the river, which may offset the impacts of salt mobilisation from wetlands and floodplains. This means that determining the salinity impacts of individual environmental activities remains a complex and open issue for which there is a paucity of modelling tools able to analyse policy options around this issue.

Jolly *et al.* (2012) recently reviewed the considerable volume of research and investigation that has taken place over the past 15–20 years on the impacts of natural and artificial flooding on salt loads to the River Murray. It was found that while lack of data can often inhibit an estimation of the salt loads generated at any individual site for a particular flood due to site heterogeneity, the driving processes are well understood. At data–rich sites such as the Chowilla floodplain it has been possible to develop highly sophisticated groundwater and hydrodynamic surface water models that provide robust estimates of salt loads produced by both natural floods and environmental watering activities. At many other sites, this level of sophistication may not be possible due to the limitations of low data availability and resources.

The review of Jolly *et al.* (2012) also highlighted that the eWater CRC and its partners have been developing a next generation integrated modelling system, known as Source IMS (Welsh *et al.*, 2012). Source IMS is a modelling environment containing algorithms and approaches that allow defensible predictions of water flow (and constituents) from catchment sources to river outlets at the sea. Source can be used in the assessment of changes to water quantity and quality due to changes in land use; natural climatic variability and climate change; changes to water demands (irrigation, urban, ecological), water management infrastructure (including dams, weirs storages and other storage reservoirs) and the impact of all of the above on various ecological indices for waterways and their surrounding floodplain areas. Given that in the near future Source IMS is likely to replace the current river system model for the River Murray, any development of decision support tools for predicting salt accessions from floodplains/wetlands should be compatible with this new modelling platform. CSIRO has developed functionality in Source IMS that allows modelling of the water fluxes between groundwater and surface water in wetlands/floodplains (Jolly *et al.*, 2010; Rassam, 2011). The review suggested that a logical extension to this work is to develop capability within Source IMS to predict the salinity accessions from wetlands/floodplains to rivers following overbank flows and environmental watering activities. This will allow salt generation processes by both groundwater and surface water to be modelled. The review recommended that a rapid assessment tool be developed within Source IMS to ensure tight coupling with the river systems model and thus allow rapid testing of potential flow delivery and watering scenarios.

2.4. Murray–Darling Basin Plan

The Murray–Darling Basin Plan has been developed to address a history of over allocation in regions of the Murray–Darling Basin. Consistent with the objectives of the Water Act 2007, the MDBA has been charged with defining the environmentally sustainable levels of take (ESLT). The ESLT is defined in the Water Act 2007 as the level at which water can be taken from that water resource which, if exceeded, would compromise:

- key environmental assets of the water resource; or
- key ecosystem functions of the water resource; or
- the productive base of the water resource; or
- key environmental outcomes for the water resource.

In defining the ESLT, the MDBA activities have included: the determination of the amount of water needed for the environment, known as the environmental water requirement (EWR), to protect, restore and provide for the ecological values and ecosystem services of the Basin; and the determination of long–term average sustainable diversion limits (SDLs), which must not compromise key environmental assets, key ecosystem functions, the productive base and key environmental outcomes for the water resource (MDBA 2012). The principle of the approach used is the ‘reinstatement of ecologically significant parts of the flow regime’ (MDBA, 2012; p. 32). Environmental water requirements are used in hydrological models to establish water demands at a gauge targeting specific aspects of the flow regime, where these demands are intended to represent the broader environmental flow requirements of river valleys and reaches (MDBA, 2011c; p. 18).

The MDBA identified over 2000 environmental assets in the Basin. Of these, 122 of these were used as hydrologic indicator sites, many of which are hydrologically connected (MDBA, 2011c; p. 32). A key assumption of the hydrologic indicator site approach and its underpinning principles is that provision of an adequate flow regime at indicator sites is representative of the environmental water requirements of the broader suite of key environmental assets and key ecosystem functions across the Basin (MDBA, 2011c; p. 33). This assumption remains untested.

The proposed Murray–Darling Basin Plan (MDBA, 2011c) anticipates sustainable diversion limits for the entire basin that, on average, will return 2,750 GL yr^{-1} from consumptive use to the environment. The 2,750 GL yr^{-1} is based on a 2009 baseline, which includes 823 GL yr^{-1} (average) acquired for the environment prior to 2009 (Table S.2.1, Schedule 2 of the Plan; MDBA, 2011c); thus the overall decrease in average consumptive use is 3,573 GL yr^{-1}. A review (CSIRO, 2011) of the earlier Guide to the Murray–Darling Basin Plan (MDBA, 2010) showed that a reduction of 3,500 GL yr^{-1} in diversions would increase flows to South Australia, but would still lead to shortfalls in very dry periods in meeting environmental water requirements in South Australia. Kirby *et al.* (2012b) showed that the combined impact of a projected median climate change and the Proposed Plan was that the increases in environmental flows under the Plan may be much reduced by climate change.

The detailed impact of the Proposed Plan on flows into South Australia is difficult to assess. For one thing, much will depend on revisions to the upstream states’ water sharing plans from 2014 (NSW) to 2019

(Victoria). For another, although a framework for environmental water use has been established (CEW, 2011a), environmental watering plans have not yet been made in detail by the Commonwealth Environmental Water Holder. Nor has it been decided whether or how much environmental water may be traded (cf. CEW, 2011b). Kirby *et al.* (2006) show that, in principle, counter–cyclical trading of environmental water could significantly increase the overall volume of water available to the environment. Such water trading may lead to significant average increases in flows, but may not be of much benefit to the environment during extended droughts (Kirby *et al.*, 2012c).

Environmental Water Requirements for South Australia

Within South Australia, hydrologic indicator sites defined by the MDBA as 'asset' hydrologic indicator sites include:

1. The River Murray channel, connected streams and wetlands;
2. The valley section floodplains (including Chowilla and other Riverland floodplains);
3. The gorge section floodplains;
4. The Coorong, Lower Lakes and Murray Mouth (CLLMM).

Detailed reports on the EWRs for the Riverland–Chowilla Floodplain and the Coorong, Lower Lakes and Murray Mouth have been published by the MDBA (http://www.mdba.gov.au/draft–basin–plan/science–draft–basin–plan/assessing–environmental–water–requirements; visited 25/06/2012).

The Riverland Chowilla represents the riverine–floodplain component of the Lower River Murray. Using (Newall *et al.* 2009) and (Overton *et al.* 2006a) the MDBA and DLWBC each established EWRs for the Riverland Chowilla, where the relationship between daily flows at the South Australian border and extent of wetting of vegetation communities was the focus. The EWRs established for Riverland–Chowilla are documented in Table A.1 contained in Appendix 2. MDBA EWRs have been used in the development of the Basin Plan (MDBA, 2011c; Table A.2) and all EWRs have been used in evaluating the Basin Plan (Pollino *et al.*, 2011; Lamontagne *et al.*, 2012). As reviewed in the predictive modelling sections, EWRs are hydrologic surrogates of condition (habitat suitability) limiting the analysis of targets to an analysis of being 'met' or 'not met' over a defined period. They give little power to understanding the likely change in the community mosaic through time as they are not spatially explicit or able to distinguish ecological responses over shorter time periods.

The Coorong Lower Lakes Murray Mouth (CLLMM) region represents the end of the Murray system. EWRs for the CLLMM region have been developed based on the salinity requirements of *Ruppia tuberosa* (tuberous sea tassel) and *R. megacarpa* (large-fruit sea tassel), salt export targets and lake levels. A hydrodynamic model (Webster 2007), a states-based ecological response model (Lester and Fairweather 2009) and biological information in (Brock 1982a) were used to define these EWRs. *Ruppia* spp. are important primary producers in the Coorong (e.g. the turions of *R. tuberosa* are an important food source for migratory waders (Paton *et al.* 2001), and may act as indicators of the overall health of the ecosystem. Knowledge gaps still remain in deriving EWRs (and salinity thresholds) for *Ruppia* spp. (in particular *R. megacarpa*, which is now absent from the Coorong and Murray Estuary). The hydrodynamic model on which the flow scenarios are based is limited to a 1-D representation, and the ecological response model does not account for transitioning between states (Lester and Fairweather 2009).

Limitations of the EWR approach and its application are given in Pollino *et al.* (2011) and Lamontagne *et al.* (2012). The major limitations were 1) lack of riverine targets that account for hydraulic connectivity and explicitly address the requirements of native fish; 2) requirements of the broader floodplain landscape between the two assets sites; and 3) limited representation of ecological complexity for evaluating the ecological outcomes of hydrologic scenarios.

Ecological outcomes under the proposed Basin Plan

There have been numerous calls to "downsize" river systems (see Pittock and Finlayson 2011; Overton and Doody 2010; Hall *et al.* 2011) by adopting a policy of re–instating a pre–regulation flood regime (timing, frequency, duration) for small magnitude floods, but not attempting to deliver medium or large floods. This approach reduces the spatial extent of the floodplain that is inundated frequently enough to maintain the character of the existing ecological communities, and therefore the volume of water required to manage the river in a more ecologically sensitive manner.

With the introduction of the Murray–Darling Basin Plan and the likelihood of lower rainfall due to climate change, CSIRO (2012) suggest that under a minimum flow scenario of 2,800 GL year^{-1} opportunities exist to generate flows between 40–80 GL day^{-1} to inundate mid level floodplains. However, these communities are unlikely to return to a healthy condition if flood frequency and duration intervals are not met. Medium elevation wetlands will receive an improvement in flood frequency and duration up to flows of 60 GL day^{-1}, but it seems likely that the overall condition of these wetlands will decline into the future and that no improvement will be seen by high elevation areas. It is very likely the future floodplain will be significantly reduced in area with floodplain vegetation transitions occurring as the ‘active floodplain’ reduces from an area inundated by flows of 140 GL day^{-1} to a reduced area that is inundated by flows between 70 and 80 GL day^{-1} (Lamontagne *et al.*, 2012).

The current approach to the recovery, and delivery of environmental water via the Basin Plan suggest that reinstating frequent small floods (<70 GL day^{-1}) is the likely management paradigm for the lower River Murray in coming decades. This approach is anticipated to have significant benefits (Lamontagne *et al.* 2012); frequent small floods function as the primary source of water sustaining lowland river floodplains in arid regions and maintain soil moisture and water levels in wetlands that increase the potential for subsequent flows to travel further downstream and/or inundate larger areas (see Leigh *et al.* 2010).

Despite the potential benefits for low–elevation sections of the floodplain and the river channel (Lamontagne *et al.* 2012), the concept of downsizing rivers overlooks the role of the interface between the higher elevation aquatic (regularly inundated) and terrestrial (never inundated) zones in subsidising terrestrial food webs. Abandonment of large sections of floodplain may create a potentially hostile (e.g. highly salinised) zone that is neither aquatic nor terrestrial. On these high elevation sections of floodplain, soil type and/or condition, or the ecological disturbance created by occasional floods may preclude the development of terrestrial communities. In areas where biota have previously moved between terrestrial (e.g. mallee) and floodplain habitats to take advantage of different sources of energy and habitat, a significant change in vegetation structure may preclude movement and generate a barrier to energy flux. It is generally considered that the flux of material from floodplain to the river is the major direction of energy movement. However, there is evidence of fluxes from the riverine landscape to the terrestrial landscape (Ballinger and Lake 2006). In arid streams, secondary production of insects contributes substantially to the food supply of insectivores, including birds, spiders and reptiles (Jackson and Fisher 1986; Lynch *et al.* 2002; Sabo and Power 2002). Faunally transported fluxes of energy may be extremely important for terrestrial foodwebs (see review by Ballinger and Lake 2006) via mobile–link organisms that move resources between aquatic and terrestrial ecosystems (Lundberg and Moberg 2003). For example:

- Nutrient export may occur as larval and nymphal stages of macro invertebrates mature in the floodplain wetlands and river channel and migrate to terrestrial environments.

- Herbivores (e.g. kangaroos) that graze on floodplain vegetation, and birds that prey upon zooplankton and fish, transport at least part of their faeces into the terrestrial environment.

3. The floodplain – river ecological communities

Hydrology is widely acknowledged as the dominant variable structuring rivers and floodplains (Tockner and Stanford 2002; Power *et al.* 1995; Walker *et al.* 1995). Flow provides critical cues and triggers in establishing an appropriate environment for ecological function (Figure 3.1). This includes provision of habitat for supporting the life cycle requirements of species, in providing lateral and longitudinal connections for migrations and 'boom' periods, as well as provision of refugia in 'bust' periods.

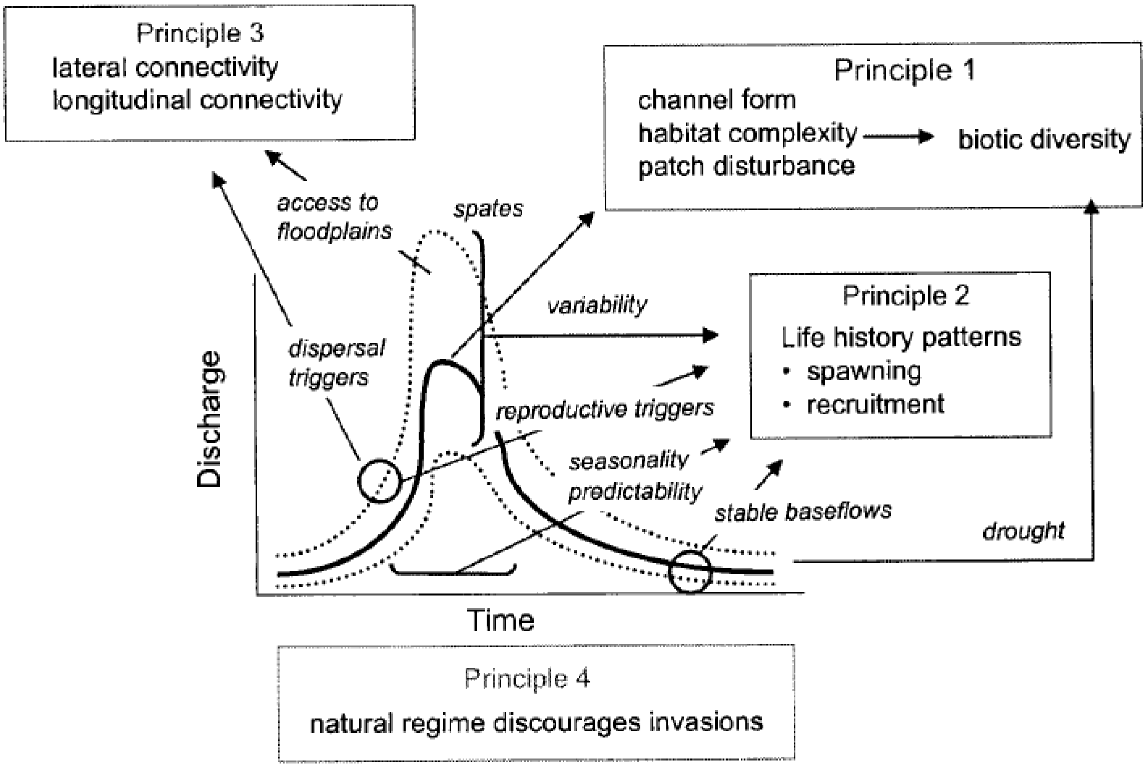

Figure 3.1 The relationship between flow regime and biodiversity in river–floodplain ecosystems. Source: Arthington and Pusey (2003).

The dynamics of lateral hydrologic river and floodplain connections promotes ecological processes through the exchange of matter and organisms (Junk *et al.* 1989). Fluvial dynamics from flooding plays a major role in maintaining a diversity of lentic, lotic and semi–aquatic habitat types, each represented by a diversity of successional stages (Ward *et al.* 1999; Amoros and Bornette 2002). Variations in the magnitude and spatial extent of flood–mediated disturbances creates habitats that are characterised by stages of succession, resulting in dynamic and diverse terrestrial and aquatic habitats (Whited *et al.* 2007), creating a diverse mosaic of vegetation across floodplains (Ward *et al.* 1999; Merritt *et al.* 2010a). Indeed, the distribution and age of river–floodplain complexes reflects a legacy of hydrologic conditions, from large scouring floods that create expansive areas for recruitment, to extended periods of moderate to low disturbance that promote succession (Whited *et al.* 2007).

Several processes and characteristics at both landscape and local levels govern the attributes and functions of floodplain communities (Nilsson and Svedmark 2002). Landscape–scale features include topography, geology, and climate (Stanford *et al.* 1996). These features set the template for an ecosystem. At a local scale, connectivity of the river and floodplain and with adjacent systems regulates the openness of the system and the frequency and magnitude of physical disturbances determine the patterns of succession and rate of species turnover (Nilsson and Svedmark 2002). Connectivity between the river, floodplain, and its interface with terrestrial systems result in structural and functional elements that result from and contribute to the spatio–temporal dynamics, the diversity and high productivity of floodplain ecosystems (Ward *et al.* 1999; Nilsson and Svedmark 2002; Opperman *et al.* 2010), where floodplains represent complex ecotones between river channels and terrestrial systems. At the finer scale, floodplains are composed of patches of habitats, and at this scale, biotic interactions dominate (Stanford *et al.* 1996).

This section describes the current state of knowledge for important river and floodplain species, functions and processes relevant to environmental flow management in the lower River Murray. It focuses on studies undertaken in the lower River Murray in the context of the international literature.

3.1. Floodplain overstorey vegetation

The native floodplain overstorey vegetation along the lower River Murray in south–eastern Australia has been in decline for several decades (Jolly *et al.* 1993). This decline is due to the introduction of river regulation (locks, dams and weirs), long term drought and abstraction for irrigation and domestic purposes, causing a reduction in flooding frequency and an increase in soil salinisation (Cunningham *et al.* 2007; Mac Nally *et al.* 2011; Overton *et al.* 2006a). Groundwater salinity and salinisation is more important in the lower River Murray than upstream as the floodplain is underlain by naturally occurring, highly saline groundwater. 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 by a number of factors including saline groundwater. They also concluded that long–term health of vegetation along the River Murray was at greatest risk from soil salinisation. Naturally saline groundwater levels have been elevated by the water ponding at locks, as well as from irrigation. Salinisation of floodplain soils has increased due to elevated watertables and increased saline groundwater discharge to semi–arid floodplain environments (Doble *et al.* 2006; Holland *et al.* 2009a). Evapotranspiration has caused salt to accumulate in the upper layers of the soil (Jolly *et al.* 1993, Overton *et al.* 2006a). Under natural conditions, flooding would leach the soil, however reduced flooding frequency and below average rainfall have prevented this. The prolonged drought has increased water stress to trees already in decline. As a result of changes to water availability, the dominant floodplain tree species are experiencing severe stress, leading to extensive tree deaths. It is estimated that over 70 % of riparian tree vegetation in wetlands and floodplains within the Murray–Darling Basin are in poor health or dead (Cunningham *et al.* 2009; MDBC 2003, 2005; Smith and Kenny 2005). Flows less than 40 GL day^{-1} and between 40–80 GL day^{-1} have become greatly reduced since regulation. These flows connect elevated floodplains and are important in transferring nutrients and materials between floodplain and channel habitats (CSIRO 2012). Alternatively, weir raising has permanently connected low level wetlands creating perennial rather than temporary wetlands. Flows greater than 80 GL day^{-1} have also been greatly reduced by upstream storage and diversion, impacting health of higher elevation temporary wetlands (Marsland *et al.* 2009; Zampatti *et al.* 2011c). The current condition of low and moderate elevation terrestrial vegetation is currently poor (Marsland *et al.* 2009; Gehrig *et al.* 2010; Zampatti *et al.* 2011c), however some recovery is noted after recent flooding (Gehrig *et al.* 2012a).

Key floodplain overstorey species in the lower River Murray–Darling Basin include river red gum (*Eucalyptus camaldulensis* var. *camaldulensis*), black box (*Eucalyptus largiflorens*), river cooba (*Acacia stenophylla*) and lignum (*Muehlenbeckia florulenta*). Tree species in particular, play an important ecosystem role providing carbon to food webs (Baldwin 1999; Briggs and Maher 1983; Colloff and Baldwin 2010) via leaf litter and habitat to a broad range of floodplain and aquatic fauna. Flood tolerant river red gum has the highest water requirement, situated adjacent river and feeder channels. Black box are more drought and salt tolerant, located further from channels in elevated floodplain locations, while river cooba occur among both red gum and black box, demonstrating both drought and salt tolerance.

Floodplain trees have access to a variety of potential water sources including lateral recharge from creeks, indirect groundwater use at the capillary fringe and freshwater lenses that sit above the saline groundwater which are recharged from flooding (Mensforth *et al.* 1994; Thorburn and Walker, 1994; Thorburn *et al.* 1994; Holland *et al.* 2006; Holland *et al* 2009b; Doody et al, 2009). All tree species are opportunistic water users and surface water–groundwater interactions determine the degree to which vegetation use groundwater, river/creek water and rainfall and are dependent on soil type, recharge rates, aquifer conductivity, groundwater depth, groundwater salinity, flooding frequency and rainfall quantity. Terrestrial vegetation possesses a range of adaptations that allow them to survive within a variable landscape as detailed in Roberts *et al.* (2012), which describes relationships between vegetation location on floodplains and specific adaptations for survival in these conditions. Black box for example, lowers transpiration during drought stress, sheds leaves to further reduce transpiration and lower stomatal conductance (Jolly and Walker 1996). They also flower after a flood irrespective of season (Cale, 2009).

A series of Murray–Darling Basin terrestrial vegetation literature reviews have been undertaken since 2000 to specifically highlight our current knowledge of vegetation water requirements, regeneration, recruitment, seed banks, age structure and phenology (Roberts and Marston 2000; George *et al.* 2005; Jensen *et al.* 2007; Jensen *et al.* 2008; Johns *et al.* 2009; Cale 2009; Roberts and Marston 2011). While significant ecological investigations have been undertaken for terrestrial vegetation to date, ecological response knowledge gaps exist, specifically related to environmental water requirements and the impact of four different flow scenarios (2,800, 3,000, 3,500, 4,000 GLyear^{-1}) proposed within the Murray–Darling Basin Authority Plan on vegetation health relative to the baseline case in South Australia (Pollino *et al.* 2011; CSIRO 2011; CSIRO 2012).

Current knowledge of vegetation water requirements

A key driver of the ecology of rivers and their associated floodplain wetlands is the flow regime (Junk *et al.* 1989; Poff *et al.* 1997; Bunn and Arthington 2002). Critical elements of the flow regime to maintain lateral connectivity between river and floodplains, longitudinal channel connectivity in a system of pools and ecological function include flood frequency, flood duration and flood timing. Of considerable importance to terrestrial vegetation is the maximum inter–flood dry period and source of water required to maintain good health and future longevity. Flood depth is important to ecological assets such as birds and fish but thought to be less important to terrestrial vegetation. Table 3.1 summarises the current water requirement knowledge for each species. Variability in water requirements is shown within each species generally and currently Roberts and Marston's (2011) "Water regime for wetland and floodplain plants" is considered the source book to refer to, however additional literature is considered in Table 3.1.

Table 3.1. Current knowledge of water requirements for river red gum, black box and river cooba in the Murray–Darling Basin. Adapted from Roberts and Marston 2011.

	Water sources	**Flood frequency (1 in X years)**	**Flood duration**	**Dry period (years)**	**Flood timing**	**Salinity tolerance**
River Red Gum	Groundwater Rainfall Flooding Ponded surface water (Mensforth *et al.* 1994; Thorburn and Walker, 1994; Thorburn *et al.* 1994; Holland *et al.* 2009b)	1 in 3 – forest (Holland *et al.* 2009b; Overton and Doody, 2007) 1 in 2–4 woodland (in Roberts and Marston, 2011) 1 in 3–5 (Wen *et al.* 2009) 1 in 2–4 (MDBC 2006a; Young 2001)	5–7 months forest / 2–4 months woodland (in Roberts and Marston, 2011) 4–7 months (Young 2001) 64 days (Wen *et al.* 2009)	5–15 months (Johns et al, 2009) Maximum 5 –7 years (Doody and Overton 2009; Cale, 2009)	Spring–summer (in Roberts and Marston, 2011; Robertson et al, 2001) Winter–summer (MDBC 2006a)	30,000 (Overton and Jolly, 2004) 40, 000 (Mensforth *et al.* 1994)
Black Box	Groundwater Rainfall Flooding (Akeroyd et al, 1998; Thorburn *et al.* 1993; Jolly and Walker, 1996; Doody et al 2009; Holland *et al.* 2006)	1 in 3–5 (MDBC 2006a; Young 2001) 1 in 3–7 (in Roberts and Marston, 2011)	3–6 months (in Roberts and Marston, 2011; Akeroyd et al, 1998)	Maximum 10 (Taylor et al, 1996) >10 (Doody and Overton, 2008)	Follow natural timing but not critical (in Roberts and Marston, 2011) Late winter–summer (MDBC 2006a)	40,000 (Jolly *et al.* 1993) 55,000 (Overton and Jolly, 2004) 60,000 Streeter *et al.* 1996)
River Cooba	Groundwater Rainfall Flooding Ponded surface water (Doody *et al.* 2009)	1 in 3 –7 (Wilson et al, 2009)	2–3 months (in Roberts and Marston, 2011; MDBC 2006a)	Maximum > 10 (Doody and Overton, 2008; in Roberts and Marston, 2011)	Late winter–summer (MDBC 2006a)	30,000 (in Roberts and Marston, 2011)

River red gum access the greatest number of water sources given their low tolerance of drought. Although flooding and ponded surface water are less important for black box and river cooba, it is important to note that both species are often found adjacent channels in the “flushed river bank zone” in mixed river red gum communities. When located within the flushed zone, lateral bank recharge and flooding is required to maintain tree health. Flooding is important for black box situated in higher floodplain locations in the lower River Murray, to replenish soil stores and flush accumulated salt from the soil (Jolly and Walker, 1996). The degree of dependence on a particular water source depends on soil hydraulic properties and a combination of soil and groundwater salinity. River red gums have been found to rely on surface water when a channel is less than 15 m away, obtaining between 30–50 % of their water from surface water. However, trees do not rely solely on surface water, accessing relatively saline groundwater while in close proximity to a permanent channel (Thorburn and Walker, 1994). Trees greater than 15 m from permanent channels rely mainly on groundwater including during times of significant stress (Thorburn and Walker, 1994). Anecdotally, on the Chowilla floodplain, many river red gums located away from permanent water bodies have died.

Opportunistic water use by black box has been interpreted as a strategy to conserve energy by extracting the source requiring the least amount of energy. For example, within 50 m of Chowilla Creek, lateral bank recharge was important to maintain black box growth and health (Jolly and Walker, 1996). Thorburn *et al.* (1993) found that black box survival between extended dry periods relied upon water extraction from moderately saline groundwater in the Chowilla region. Doody *et al.* (2009) demonstrated that transpiration rates of both river cooba and black box were very conservative after extended drought in a saline environment. Both trees obtained water from rainfall when possible, and otherwise from groundwater, unless in the vicinity of the bank flushed zone.

Annual return intervals or the flood frequency required to maintain terrestrial vegetation health is variable for red gum and black box (Table 3.1). Return periods of between 1 to 5 and 1 to 8 years are suggested for red gum and black box respectively, after which tree health declines. Doody and Overton (2009) found that after prolonged drought stress (greater than 5 years without flooding), red gum not only minimised their canopy but reduced their sapwood area, potentially reducing their ability to respond and recover when water availability increased.

More recently, in the absence of overbank flooding, flood pulses have become more important to sustain vegetation health. Raising water levels has been shown to improved canopy health in extremely stressed river red gums (Holland *et al.* 2009b; Souter *et al.* 2013). It is suggested that pulse flows should occur every 9–18 months to support health of trees situated in the lateral recharge zone of river channels (Souter *et al.* 2013). Remote sensing also identified that raised water levels along the River Murray in South Australia from the delivery of irrigation water during summer improved river red gum health within 60-90 m of the river channel (T. Doody, pers. comm.). The observed improvement in river red gum health within 60-90 m due to summer irrigation flows is within the range (50-200 m) of lateral bank recharge estimated in previous studies in the lower River Murray (Holland *et al.* 2006; Doody *et al.* 2009; Holland *et al.* 2009b; Holland *et al.* 2013).

The amount of time a flood persists also varies (Table 3.1). River red gum are considered to be more flood tolerant as indicated by their position in the landscape and can tolerant flooding for up to two years and even four years if good health prevailed initially (Roberts and Marston, 2011). Heinrich (1990, cited in Roberts and Marston, 2011) and McEvoy (1992) found reduced transpiration and growth of black box seedlings after 30 and 70 days and 62 days respectively of continual flooding, demonstrating that black box seedlings preferred less frequently flooded parts of the floodplain. In contrast, mature blackbox did not have reduced transpiration rates following 78 days of inundation (Akeroyd *et al.* 1998). Robertson *et al.* (2001) found that one large flood (equivalent to one in every three years) which persisted for a longer duration was comparable in terms of wood production to short spring floods every year, indicating that flood duration plays an important part in vegetation maintenance. They also found that wood production was significantly greater for trees flooded each year in spring and summer or summer (Robertson *et al.* 2001).

While floodplain systems require both dry and wet periods to function optimally, extended periods of drought lead to vegetation health decline as witnessed over the period from 1997–2010. The optimal period of dryness is linked to flood frequency and only mentioned for river red gums in the literature (Johns *et al.* 2009). After five to seven years without flooding (and with low rainfall) severe degradation was obvious (Table 3.1). Black box and river cooba have all survived more than 10 years without significant rainfall or flooding, however health is poor (Table 3.1) (Roberts and Marston 2011).

In defining standard EWR's for floodplain vegetation, there is an implicit assumption that it is just a lack of water *per se* that controls vegetation health and so it is assumed that the restoration of flows to something close to the natural flow requirements will result in recovery of the vegetation communities. This assumption does not factor in the current poor health condition of many of the vegetation communities of the lower River Murray. Nor does it account for important anthropogenic factors such as high soil salinity which have been shown by numerous studies over the last 20 years to be a very important influence on floodplain vegetation health and regeneration. The effect of high salt content in saline soils is similar to drought conditions in non–saline soils as the ability of plants to extract water is reduced (Munns and Termaat 1986). The physiological tolerance of the floodplain tree species is approximately –2.5 MPa for *E. camaldulensis* (Mensforth *et al.* 1994; Holland *et al.* 2009a; Doody *et al.* 2009) and –4.0 MPa for *A.*

stenophylla and *E. largiflorens* (Doody *et al.* 2009; Miller *et al.* 2003; Holland *et al.* 2009b; Holland *et al.* 2006; Bramley *et al.* 2003; Zubrinich *et al.* 2000). This equates to maximum soil chloride values of approximately 20 g Cl L^{-1} and 30 g Cl L^{-1} and groundwater salinity values of approximately 35 g L^{-1} TDS and 50 g L^{-1} TDS, respectively.

Recruitment and regeneration

Understanding of the water requirements to prompt recruitment and regeneration are less understood for floodplain overstorey vegetation species (Table 3.2). It appears unlikely that any of the four dominant overstorey species form persistent seed banks (Roberts and Marston 2011) with red gum and black box forming aerial seed banks, held in the canopy for up to two years (Jensen *et al.* 2007, 2008). The extent of aerial seed banks is highly dependent on environmental conditions over the previous 12 months (Jensen *et al.* 2007). Germination of seed is triggered by high rainfall and receding water levels after flooding, with the dominant overstorey species germinating as flood water recede, because they cannot germinate when submerged (George *et al.* 2005; Roberts and Marston 2011; Jensen, 2008). Like most of the Mimosaceae, it is likely that river cooba seeds have a physical dormancy that requires scarification of the seed coat before germination. Large scale river red gum and black box recruitment occurs after flooding, with seedlings establishing within the flood strand lines (Margules *et al.* 1990, Jensen *et al.* 2008) forming an age specific band related to each flood. Artificial watering is known to promote river red gum germination (Jensen *et al.* 2008) although not to the level of natural flooding. Investigations by Capon *et al.* (2009) show lignum seedlings are more tolerant of drying than flooding, but can also remain dormant for extended periods, growing new branches from the rootstock in response to increased water availability (Roberts and Marston, 2011). Soil conditions in the years following germination are critical for survival (George 2004). Although adequate water is required to stimulate growth, black box seedlings are not flood tolerant (Heinrich 1990, cited in Roberts and Marston, 2011), whereas river red gum seedlings require sufficient water in the first two to three years while a deep tap root develops (George 2004).

Recent recruitment of black box has declined and is not enough to ensure population survival in the lower River Murray without increasing current flooding frequencies and high salinity levels (George *et al.* 2005). Jensen *et al.* (2007) suggest environmental water could be managed to support seedling growth after germination in both river red gum and black box, particularly when only short periods of 4 to 6 weeks are required after germination (Table 3.2).

Table 3.2. Critical factors affecting recruitment and regeneration of red gum, black box and river cooba.

	Flood timing– regeneration	**Follow–up flood (years)**	**Depth of flood seedling establishment**	**Flood duration (weeks)**
River Red Gum	Recession Spring/early summer Dexter (1978)	Spring germination – Summer follow–up (Jensen *et al.*, 2008)	20–30 cm (in Roberts and Marston, 2011) 50 cm (MFAT, 2009)	4–6 weeks (in Roberts and Marston, 2011)
Black Box	Recession Spring – Summer Dexter (1978)	Summer after germination (in Roberts and Marston, 2011)	4 cm (McEvoy, 1992) 30 cm for 1 year old (Young et al, 2003)	4 weeks after 2 months age (Heinrich, 1990, cited in Roberts and Marston, 2011).
River Cooba	Unknown. Seed fall from Spring to summer (in Roberts and Marston, 2011)	Unknown.	Unknown.	Unknown.

3.2. Aquatic and floodplain understorey vegetation

A total of 435 species of herbs, forbs, grasses and shrubs (including nine species listed as rare, one species listed as endangered in South Australia; and 153 exotics, 17 of which are declared noxious in South Australia with six being weeds of national significance) have been recorded in the South Australian River Murray Corridor (main channel, floodplain, associated wetlands, Lower Lakes and Coorong) since 2000 (Appendix 3). Of the 281 native understorey species that have been recorded, 184 are associated with aquatic or floodplain environments and are not found in the adjacent terrestrial ecosystems (Jessop and Tolken 1986; Cunningham *et al.* 1992; Jessop *et al.* 2006). Therefore, the River Murray Corridor provides the only suitable conditions for persistence of these species in the region. Hence, similar to other aquatic and floodplain systems, they contribute to regional biodiversity by providing habitat for different species that are not present in other systems (sensu Sabo *et al.* 2005).

The flora of the permanently inundated areas is largely composed of aquatic and emergent species, most of which have cosmopolitan distributions and are found on all continents except Antarctica (e.g. *Typha domingensis, Potamogeton pectinatus*) (Sainty and Jacobs 2003). In contrast, the flora of the River Murray floodplain is very different from systems that have regular flooding cycles. Often the species present in systems with regular flooding cycles are perennial emergent or amphibious species; whereas, species present on the River Murray Floodplain are often short lived annuals (examples of Grime (1979) r–selected species). Nicol (2004) suggested that the majority of plant species present on the Darling River floodplain (many of which are present on the lower River Murray floodplain) are more similar to desert annuals than aquatic plants.

Walker and Thoms (1993) stated that changes to hydrology change the habitat template and that the physical, chemical and biological character of rivers and wetlands will change in response to the altered hydrology. The changes in hydrology of the lower River Murray, brought about by river regulation and abstraction (such as stable water levels for extended periods, reduced flows, less frequent small to medium sized floods and reduced frequency and duration of overbank flows (refer Maheshwari et al. 1995; Chapter 1) have had serious consequences for plant communities. Hydrology is the primary determinant of plant communities in aquatic and floodplain systems (Mitsch and Gosselink 1993) and in combination with elevation determines the water regime experienced by individuals, populations and the entire plant community. Adaptations to the natural water regime involve life history strategies and physiological and morphological adaptations (Bunn and Arthington 2002; Lytle and Poff 2004). In addition, for each mode of adaptation, different components of the water regime appear to be relevant. For example, timing is important for life history strategies and water depth and flood duration for morphological and physiological adaptations (Lytle and Poff 2004). Due to different modes of adaptation, different species show markedly different responses to the same water regime modifications (Lytle and Poff 2004).

A consequence of stable water levels due to river regulation and the drought between 1996 and 2010, was the narrowing of the littoral zones of permanent wetlands, although they may vary due to wind (especially downstream of Lock 1) or river operations (generally in weir tail waters). The vegetation over the last ten years in the main channel was dominated by emergent species that are adapted to static water levels such as willows (*Salix* spp.), cumbungi (*Typha* spp.), common reed (*Phragmites australis*) and river club rush (*Schoenoplectus validus*) (Nicol *et al.* 2010a). Submergent species including ribbon weed (*Vallisneria australis*), curly pondweed (*Potamogeton crispus*) and milfoils (*Myriophyllum* spp.) were restricted to shallow areas by unstable sediments and high turbidity (Nicol *et al.* 2010a). Permanent wetlands are shallower, more stable habitats and often have higher species richness (e.g. Nicol *et al.* 2006; Nicol 2010), particularly of submergent species.

An additional consequence of the reduced littoral zone is the restricted habitat for species that are adapted to wetting and drying. A vegetation survey of four weir pools in the lower River Murray by Blanch *et al.* (2000) in 1994 showed that there was a 'cone-shaped' longitudinal distribution of most species (i.e. they occurred at elevations ranging from pool level to 4 – 6 m above pool in the upper 10% of the weir pool and from pool level to 1 – 1.5 m above pool level in the lower 10%). These surveys were undertaken in the year following two large overbank floods in 1992 and 1993, which provided a large area for colonisation of amphibious and floodplain species (Blanch *et al.* 2000). In addition, Blanch *et al.* (2000) reported that *Typha domingensis, Schoenoplectus validus* and *Vallisneria australis* (species adapted to stable water levels (Blanch *et al.* 1999) were restricted to lower parts of the weir pools where water levels were least variable.

During the drought, water level fluctuations were greatly reduced along the entire river's length and there were minimal overbank flows. This reduced the area available for colonisation by amphibious and floodplain species (e.g. *Alternanthera denticulata, Bolboschoenus medianus, Euphorbia drummondii, Eleocharis acuta, Cyperus exaltatus, Cyperus gymnocaulos, Xanthium* spp. and *Sporobolus mitchelli*), which were restricted to the fringes of permanent water bodies and extended no more than 1 m above pool level during the drought (Nicol *et al.* 2006; Marsland and Nicol 2007; Marsland and Nicol 2008; Zampatti *et al.* 2011). Similarly, Blanch *et al.* (2000) observed that germination and growth of these species was reduced when water levels are relatively stable, such as in the lower sections of the weir pools, immediately upstream of the lock and weir. Many of the species restricted to the lower sections of weir pools (e.g. *Schoenoplectus validus, Vallisneria americana, Typha domingensis* in Blanch *et al.* 2000) had colonised the mid reaches of weir pools during the drought (Marsland *et al.* 2010). This illustrates that the aquatic and understorey floodplain vegetation community is highly dynamic and strongly influenced by antecedent flow conditions. Vegetation surveys after a weir raising to extend a small environmental flow (32 GL day^{-1}) during the drought found that flood-tolerant and flood-dependent species (e.g. *S. mitchelli*) grew and germinated, flood-intolerant species (e.g. *A. vesicaria*) senesced, and that no aquatic plants germinated or established (Siebentritt *et al.* 2004). They suggested that the value of short term environmental flows was in maintain existing communities rather than restoring degraded communities. After the 2010/11 flood, *Alternanthera denticulata, Euphorbia drummondii, Eleocharis acuta, Cyperus gymnocaulos, Xanthium* spp. and *Sporobolus mitchelli* recruited in areas greater than 2 m above pool level (Gehrig *et al.* 2012), demonstrating the restorative effects of a large natural flood and resilience of the aquatic and understorey floodplain vegetation of the lower River Murray.

Lignum (*Muehlenbeckia florulenta*) is both drought and salt tolerant but highly reliant on flooding to drive growth, meaning that it's distribution is closely linked to flood frequency and therefore surface elevation. Lignum biomass is strongly linked to flood frequency, with vigorous growth maintained when flooded every three years. Very poor health and small shrubs prevail after seven years (Craig *et al.* 1991). Lignum forms extensive shrub lands on heavy clay soils and also occurs as understorey in floodplain woodlands (Roberts and Marston 2011). Lignum is often leafless during dry periods, but maintains vegetative growth by stem layering, and responds rapidly to rainfall or flooding by production of shoots, leaves and flowers (Chong and Walker 2005; Roberts and Marston 2011). Field studies of lignum regeneration found that seeds did not persist in the soil or mother plant and concluded that the persistence of lignum appears to depend on its capacity to tolerate drought, maintain vegetative growth and respond quickly to watering (Chong and Walker 2005).

Plant functional groups based on water regime preferences

Due to the large number of species present in the South Australian Riverine Corridor, species are classified into functional groups (based on water regime preferences and water requirements) (Table 3.3). The position each group occupies in relation to flooding depth and duration is outlined in Figure 3.2. The functional classification was based on the classification framework devised by Brock and Casanova (1997) and Casanova (2011), which was based on species from wetlands in the New England Tablelands region of New South Wales and modified by Gehrig *et al.* (2011b) to suit the South Australian River Murray. A list of species present in the South Australian Riverine Corridor and their functional groups is presented in Appendix 3.

Table 3.3: Functional classification of plant species based on water regime preferences. Modified from Brock and Casanova (1997) (*denotes exotic species).

Functional Group	Water Regime Preference	Example species
Terrestrial dry	Will not tolerate inundation and tolerates low soil moisture for extended periods.	*Atriplex vesicaria* *Maireana sedifolia* *Sclerolaena divaricata*
Terrestrial damp	Will tolerate inundation for short periods (<2 weeks) but require high soil moisture throughout their life cycle.	*Conyza bonariensis** *Carduus tenuiflorus** *Chenopodium glaucum**
Floodplain	Temporary inundation, plants germinate on newly exposed soil after flooding but not in response to rainfall.	*Lachnagrostis filiformis* *Epaltes australis* *Glinus lotoides*
Amphibious fluctuation tolerator–emergent	Fluctuating water levels, plants do not respond morphologically to flooding and drying and will tolerate short–term complete submergence (<2 weeks).	*Cyperus gymnocaulos* *Juncus usitatus* *Schoenoplectus pungens*
Amphibious fluctuation tolerator–woody	Fluctuating water levels, plants do not respond morphologically to flooding and drying and are large perennial woody species.	*Melaleuca halmaturorum* *Muehlenbeckia florulenta*
Amphibious fluctuation tolerator–low growing	Fluctuating water levels, plants do not respond morphologically to flooding and drying and are generally small herbaceous species.	*Isolepis producta* *Isolepis platycarpa* *Cyperus difformis*
Amphibious fluctuation responder–plastic	Fluctuating water levels, plants respond morphologically to flooding and drying (e.g. increasing above to below ground biomass ratios when flooded).	*Persicaria lapathifolium* *Ludwigia peploides* *Cotula coronopifolia* *Hydrocotyle verticillata*
Floating	Static or fluctuating water levels, plants respond to fluctuating water levels by having some or all organs floating on the water surface. Most species require permanent water to survive.	*Azolla* spp. *Lemna* spp.
Submergent r–selected	Temporary wetlands that hold water for longer than 4 months.	*Ruppia tuberosa* *Ruppia polycarpa* *Chara* sp.
Emergent	Static shallow water <1 m or permanently saturated soil.	*Typha* spp. *Phragmites australis* *Schoenoplectus validus*
Submergent k–selected	Permanent water.	*Vallisneria spiralis* var. *americana* *Ruppia megacarpa* *Potamogeton crispus*

The use of a functional group approach has several advantages compared to a species or community based approach:

- species with similar water regime preferences (or water requirements) are grouped together, which simplifies systems with high species richness (especially where there are large numbers of species with similar water regime preferences),
- predictions about the response of the plant community are made based on processes and does not require prior biological knowledge of the system,
- it is transferrable between systems,

- robust and testable models that predict the response of a system to an intervention or natural event can be constructed, which can in turn be used as hypotheses for monitoring programs.

However there are limitations to the approach, which include:

- loss of information on species or communities (especially if there are species or communities of conservation significance or there is a pest plant problem),
- uncertainty regarding which species should be classified into which functional group,
- important factors (e.g. salinity) are often not taken into consideration (additional factors can be included; however, this can often complicate the functional classification and in systems where there is low species richness the number of groups may be greater than the number of species).

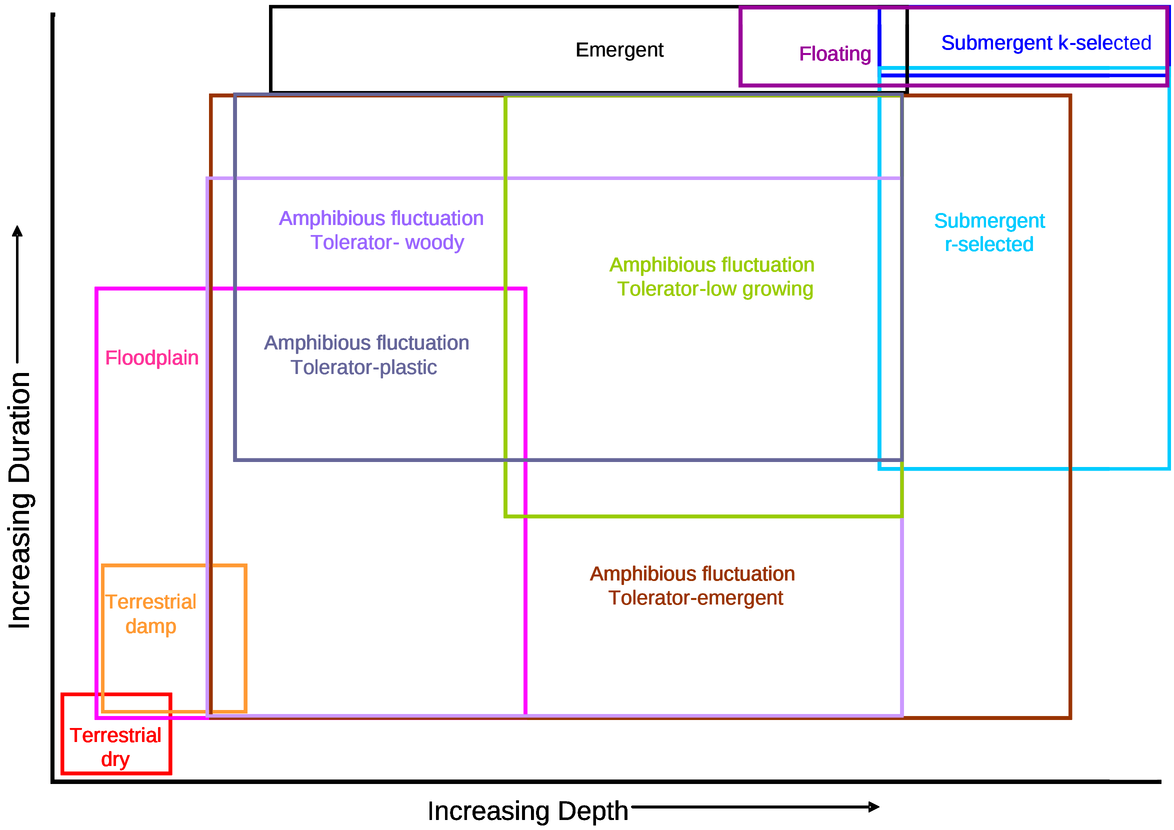

Figure 3.2 Plant functional groups in relation to depth and duration of flooding. Source: Gehrig *et al.* 2011.

The “terrestrial dry” functional group is intolerant of flooding and taxa will persist in environments with low soil moisture (Table 3.3, Figure 3.2) (Brock and Casanova 1997). Taxa from this functional group often invade wetlands that have been dried for an extended period or floodplains where there has been a lack of flooding but are generally restricted to elevated areas that never or rarely flood (Brock and Casanova 1997).

Taxa in the “terrestrial damp” group will tolerate inundation for short periods and require high soil moisture to complete their life cycle (Table 3.3, Figure 3.2) (Brock and Casanova 1997). Taxa from this functional group are often winter annuals, perennial species that grow around the edges of permanent water bodies where there is high soil moisture or species that colonise wetlands shortly after they are drawn down and riparian zones and floodplains shortly after flood waters recede (Brock and Casanova 1997).

Taxa in the “floodplain” functional group exhibit most of the traits of terrestrial species; they are generally intolerant of long–term inundation but are restricted to areas that flood periodically (they are absent from the highlands) because they only germinate after flood waters recede or wetlands are drawn down, not in response to rainfall; hence, they require flooding to regenerate (Table 3.3, Figure 3.2) (Nicol 2004). Taxa from this functional group colonise floodplains and riparian zones after flood waters have receded and when wetlands are dried (Nicol 2004). Floodplain species often have flexible life history strategies, they grow whilst soil moisture is high and flower and set seed (after which most species die) in response to low soil moisture (Nicol 2004).

The “amphibious fluctuation tolerator–emergent” group consists mainly of emergent sedges and rushes that prefer high soil moisture or shallow water but require their photosynthetic parts to be emergent, although many will often tolerate short–term submergence (Table 3.3, Figure 3.2) (Brock and Casanova 1997). Taxa from this group are often found on the edges of permanent water bodies, in seasonal and temporary wetlands, in riparian zones and areas that frequently wet and dry.

Species in the ”amphibious fluctuation tolerator–woody” group have similar water regime preferences to the amphibious fluctuation tolerator–emergent group (Table 3.3, Figure 3.2) and consist of woody perennial species (Brock and Casanova 1997). Plants generally require high soil moisture in the root zone but there are several species that are tolerant of desiccation for extended periods (Roberts and Marston 2000). Species in this functional group are generally found on the edges of permanent water bodies, in seasonal and temporary wetlands, in riparian zones and areas that frequently wet and dry.

The “amphibious fluctuation tolerator–low growing” group have similar water regime preferences to the amphibious fluctuation tolerator–emergent and amphibious fluctuation tolerator–woody groups (Table 3.3, Figure 3.2); however, some species can grow totally submerged except during flowering (when there is usually a requirement for a dry phase) (Brock and Casanova 1997). Species in this functional group are generally found on the edges of permanent water bodies, in seasonal and temporary wetlands, in riparian zones and areas that frequently wet and dry but species are usually less desiccation tolerant than species in the other amphibious tolerator groups (Figure 3.2).

The “amphibious fluctuation responder–plastic” group occupies a similar zone to the amphibious fluctuation tolerator–low growing group; except that they have a physical response to water level changes such as rapid shoot elongation or a change in leaf type (Brock and Casanova 1997). They can persist on damp and drying ground because of their morphological flexibility but can flower even if the site does not dry out. They occupy a slightly deeper/wet for longer area than the amphibious fluctuation tolerator–low growing group (Table 3.3, Figure 3.2).

Species in the “floating” functional group float on the top of the water (often unattached to the sediment) with the majority of species requiring the presence of free water of some depth year round; although, some species can survive and complete their life cycle stranded on mud (Table 3.3, Figure 3.2) (Brock and Casanova 1997). Taxa in this group are usually found in permanent water bodies, often forming large floating mats upstream of barriers (e.g. weirs), in lentic water bodies and slack waters.

“Submergent r–selected” species colonise recently flooded areas (Table 3.3, Figure 3.2) and show many of the attributes of Grime’s (1979) r–selected (ruderal) species, which are adapted to periodic disturbances. Many require drying to stimulate germination; they frequently complete their life cycle quickly and die off naturally. They persist via a dormant, long–lived bank of seeds, spores or asexual propagules (e.g. *Ruppia tuberosa* and *Ruppia polycarpa* turions in the sediment) (Brock 1982b). They prefer habitats that are annually flooded to a depth of more than 10cm but can persist as dormant propagules for a number of years allowing colonisation of temporary or ephemeral wetlands).

The “emergent” group consists of taxa that require permanent shallow water or a permanently saturated root zone, but have emergent leaves or stems (Table 3.3, Figure 3.2). They are often found on the edges of permanent waterbodies and in permanent water up to 2 m deep (depending on species) or in areas where there are shallow water tables (Roberts and Marston 2011).

“Submergent k–selected” species require permanent water greater than 10 cm deep for more than a year to either germinate or reach sufficient biomass to start reproducing (Table 3.3, Figure 3.2) (Roberts and Marston 2011). Species in this group show many of the attributes of Grime’s (1979) k–selected (competitor) species that are adapted to stable environments and are only found in permanent water bodies. The depth of colonisation of submergent k–selected species is dependent on photosynthetic efficiency and water clarity (*sensu* Spence 1982).

The Living Murray (TLM) targets and management actions have been assessed using the functional approach (e.g. Casanova 2011; Gehrig *et al.* 2012b; Gehrig *et al.* 2011b; Nicol *et al.* 2010b). For example, Nicol *et al.* (2010b) compared the species and functional approaches when assessing the benefit of watering temporary wetlands. The changes in the abundance of functional groups across multiple sites showed very clear patterns before and after watering (Figure 3.3a). A similarity of 20% cluster analysis (based on the abundance of functional groups) identified two significantly different groups that corresponded to before (red labels) and after watering (blue labels). In contrast, cluster analysis shows four groups at a similarity of 20% comparing the floristic composition of the same wetlands, with no clear groups separating the pre and post–watering surveys (Figure 3.3b).

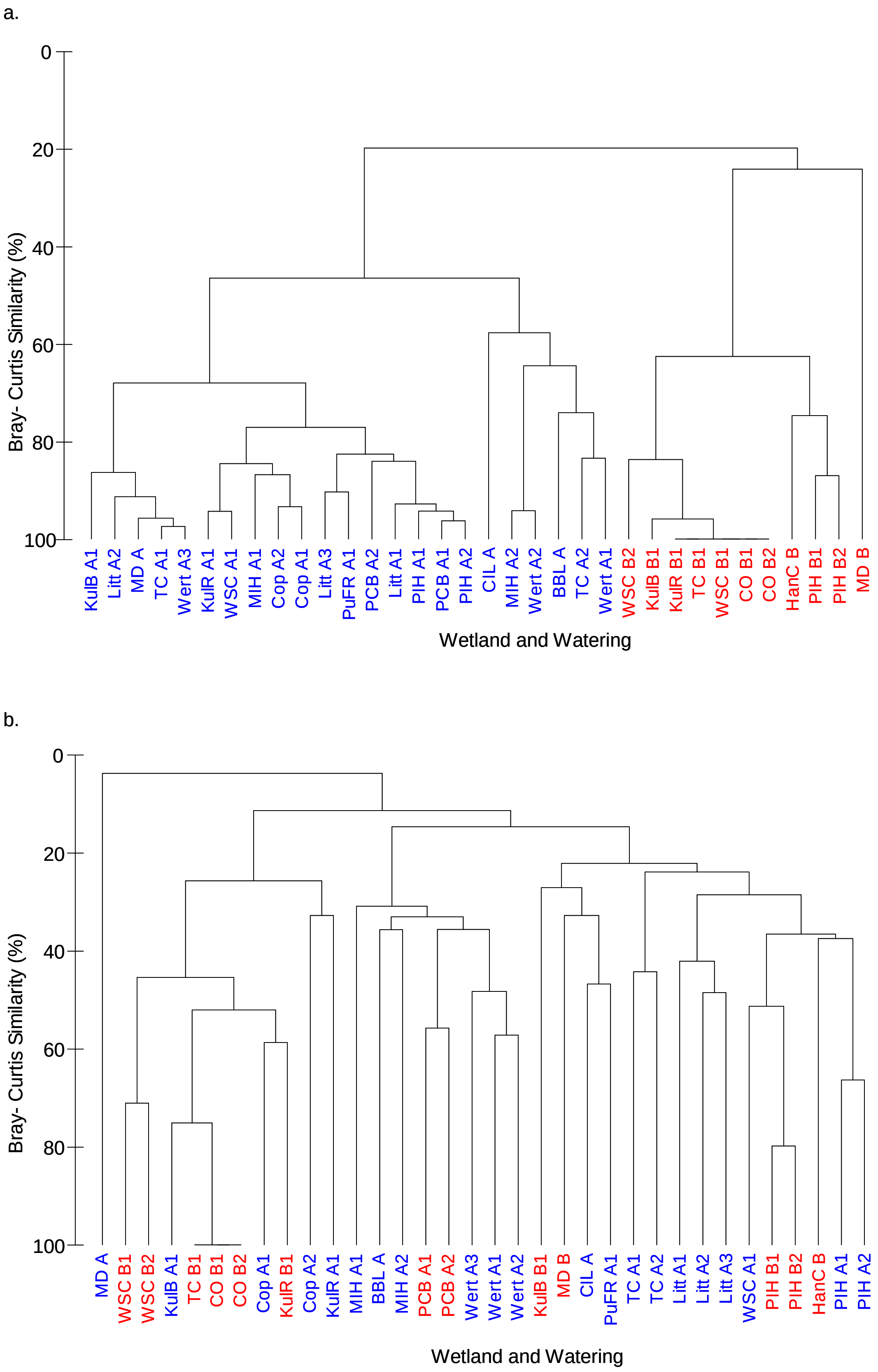

Figure 3.3 Group average cluster analysis comparing (a) the abundance of functional groups and (b) the floristic composition before (red labels) and after (blue labels) watering. Source: Nicol *et al.* 2010b.

Current knowledge of understorey vegetation dynamics in the South Australian Riverine Corridor

The vast majority of information collected on floodplain understorey and aquatic vegetation in the past 10 years has focused on the changes in plant communities through time, in response to management actions or drought and flooding. Condition and intervention monitoring undertaken as part of The Living Murray Initiative (TLM) has produced medium–term datasets, which document the change in plant communities through time on the Chowilla Floodplain (Gehrig *et al.* 2012a), Lower Lakes (Gehrig *et al.* 2012b) and Coorong (Nicol 2007; Paton and Rogers 2008). In addition, TLM intervention monitoring has documented the changes in plant communities in response to watering temporary wetlands on the Chowilla Floodplain (Nicol *et al.* 2010b) and Lower Lakes (Paton and Bailey 2010) and the response of plant communities to low water levels in wetlands downstream of Lock 1 (Australian Water Environments 2008; Nicol 2010).

In addition to TLM monitoring, several one off surveys were undertaken at numerous sites throughout the riverine corridor between 2004 and 2011 (Holt *et al.* 2005; Nicol *et al.* 2006; Weedon *et al.* 2006; Marsland and Nicol 2007; Marsland and Nicol 2008a; Marsland and Nicol 2008b; Marsland and Nicol 2009a; Marsland and Nicol 2009b; Frahn *et al.* 2012), plant communities were mapped between the Clayton regulator and Goolwa Barrage (Gehrig *et al.* 2011a), an investigation of the vegetation dynamics of the Chowilla system in permanent wetlands and temporary wetlands inundated by flows of less than 60 GL day^{-1} (Zampatti *et al.* 2011c), seed bank assessments of Lower Lakes wetlands (Nicol and Ward 2010a; Nicol and Ward 2010b) and fish habitat assessments of the main channel (Marsland *et al.* 2010), Chowilla Anabranch system (Zampatti *et al.* 2011c), Pike Anabranch system (Beyer *et al.* 2011) and Katarapko Anabranch system (Leigh *et al.* 2012) were undertaken.

With the exception of the seed bank assessments and habitat surveys, patterns of the extant vegetation and the changes in patterns through time were the primary focus of the aforementioned studies. Generally the only process investigated was the relationship with the antecedent and current surface water hydrology and salinity. Throughout the riverine corridor upstream of the barrages there were predictable changes to the understorey vegetation in response to hydrology. Downstream of Lock 1 from 2007 to 2010 there was an increase in terrestrial taxa and corresponding decrease in submergent, emergent and amphibious taxa in wetlands and around the shorelines of lakes Alexandrina and Albert as water levels fell below pool level (Nicol 2010; Gehrig *et al.* 2012b). After water levels were reinstated in winter 2010 there was an increase in the abundance of emergent, amphibious and submergent taxa and corresponding decrease in terrestrial taxa in the Lower Lakes (especially shoreline wetlands in Lake Alexandrina); however, as of autumn 2012 there were still several submergent species that were present prior to 2007 that have not been recorded (Gehrig *et al.* 2012b). Upstream of Wellington, wetlands have refilled but there has only been an increase in the abundance of emergent taxa, with submergent species largely being absent (J. Nicol unpublished data). The lack of establishment of submergent taxa is probably due to an extended period of high water levels that resulted in light levels that were insufficient for germination and survival in areas that historically supported these species. Upstream of Lock 1 (where water levels remained at pool level during the recent drought) there has also been a decrease in submergent vegetation in the main channel and wetlands since spring 2010 (J. Nicol pers. obs.).

Downstream of the barrages there was an increase of salinity in the North and South Lagoons of the Coorong from 2000 to 2010. During this period *Ruppia megacarpa* was completely lost from the North Lagoon and Murray Estuary, including the propagule bank (Nicol 2007). *Ruppia tuberosa* declined in abundance (extant vegetation and the propagule bank) in the South Lagoon and colonized the southern end of the North Lagoon over the same period (Paton and Rogers 2008; Brookes *et al.* 2009b). The change in distribution and abundance was due to elevated salinity throughout the Coorong, which exceeded 150 ‰ TDS in the southern end of the South Lagoon (Brookes *et al.* 2009b). Salinity decreased throughout the Coorong after barrages were opened due to the 2010–11 flood and *Ruppia tuberosa* germinated in the South Lagoon in winter/spring 2011 in areas where it was recorded prior to 2000 but was absent during the drought (Frahn *et al.* 2012). However, water levels in the South Lagoon fell in early January 2012 stranding the plants and preventing them from completing their life cycle and replenishing the propagule bank (Frahn *et al.* 2012).

Areas of the floodplain above normal pool level prior to spring 2010 showed signs of severe stress. Generally species richness was low (except in areas where there were interventions) and the plant community was dominated by drought tolerant terrestrial and salt tolerant taxa (Marsland and Nicol 2009a; Marsland 2010; Gehrig *et al.* 2012a). However, if an area of floodplain or temporary wetland was watered amphibious and floodplain species increased in abundance and there was a corresponding decrease of terrestrial species and area of base soil (Nicol *et al.* 2010b; Gehrig *et al.* 2012a). Figure 3.4 shows the change in understorey species richness from 2006 to 2011 on the Chowilla Floodplain. Generally species richness decreased from 2006 to 2009 with the exception of the 2007 survey where two large areas of floodplain (total area 780 ha) were watered in spring 2006 (Figure 3.4). These two areas were watered for a second time in spring 2009, which resulted in a further increase in species richness in 2010 (Figure 3.4). Areas not watered declined in species richness until the 2011 survey when there was a large increase in species richness due to the 2010–11 floods (Figure 3.4).

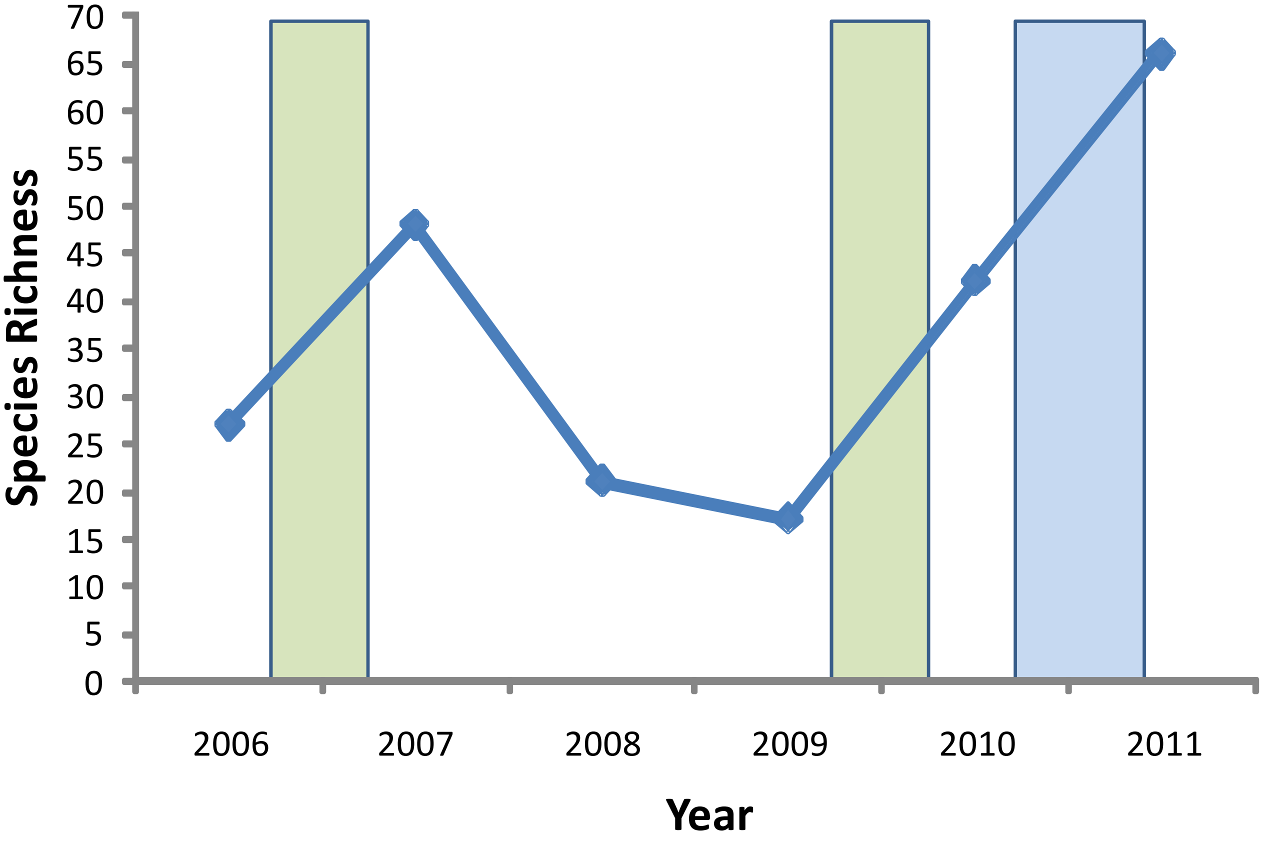

Figure 3.4 Change in understorey species richness from 2006 to 2011 on the Chowilla Floodplain (green shading represents watering and blue shading represents natural flooding). Source: Gehrig *et al.* 2012a.

Chowilla was the recipient of 27,417 ML of environmental water between 2006 and 2010 that was pumped or gravity fed into temporary wetlands, covering an area of 2,175 ha. These interventions provided conditions suitable for the regeneration of amphibious and floodplain species in the absence of flooding. However, monitoring results from Chowilla showed that areas of floodplain that had not been inundated for over ten years responded to flooding with the plant community being dominated by amphibious and floodplain species (Gehrig *et al.* 2012a). In addition, the Pike Floodplain (which was not watered during the same period) showed a similar response to Chowilla (Holland *et al.* 2013).

The response of the floodplain understorey plant community to watering or flooding was generally short lived (Nicol *et al.* 2010b; Gehrig *et al.* 2012a). After 12 months of a floodplain or temporary wetland drying, most of the floodplain and amphibious species that recruited in response to inundation had completed their life cycle and died and were often replaced by terrestrial species (Nicol *et al.* 2010b; Gehrig *et al.* 2012a). However, due to the flexible (and often short) life cycles of many floodplain and amphibious taxa the seed bank had been replenished (Nicol 2004) and there was a source of propagules for recolonisation after the next flood or watering.

Based on results from watering temporary wetlands and floodplains and the small in–channel flow pulse that occurred in spring 2005, Nicol *et al.* (2010a) proposed a state and transition model of floodplain vegetation succession. The model consists of four states, flood dependent and amphibious vegetation (state 1), terrestrial dry species (state 2), salt tolerant species (state 3) and bare soil (state 4) (Figure 3.5). State 1 is an area of floodplain that was inundated in the previous 12 months or the pool level littoral zone that has permanently high soil moisture; amphibious and flood dependent species are the dominant taxa (Figure 3.5). If the floodplain is not inundated the flood dependent and amphibious taxa will be replaced by drought tolerant terrestrial species (state 2) (*sensu* Hassam 2007; Gehrig *et al.* 2012a; Nicol *et al.* 2010b) (Figure 3.5). When an area of floodplain is dominated by terrestrial species and is inundated, the terrestrial species will be replaced by amphibious and flood dependent taxa. However, if the area of floodplain is not inundated and is in an area that has 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) (Figure 3.5). There is evidence to suggest that the switch from drought tolerant to salt tolerant species occurs when the soil salinity exceeds 20,000 µS.cm^{-1}. *Atriplex* spp. are generally not present in areas where the soil is greater than 20,000 µS.cm^{-1} (Hassam 2007) and *E. camaldulensis* trees show signs of extreme stress (Bailey *et al.* 2002). When the floodplain is dominated by salt tolerant species or bare soil, inundation of the sediment may not result in the return of flood dependent or amphibious species (*sensu* Zampatti *et al.* 2011c). The sediment may need to be inundated several times or for an extended period to allow flushing of the salt from the soil profile before the soil salinity is sufficiently low to allow the establishment of flood dependent and amphibious species. In addition, the lack of flooding may have resulted in the decline of the seed banks of amphibious and flood dependent species. Therefore, it is important that large areas of floodplain do not become dominated by salt tolerant species or bare soil because it may require a greater effort and larger amount of water to return the floodplain back to state 1 where it is dominated by flood dependent and amphibious species compared to when the floodplain is dominated by terrestrial species.

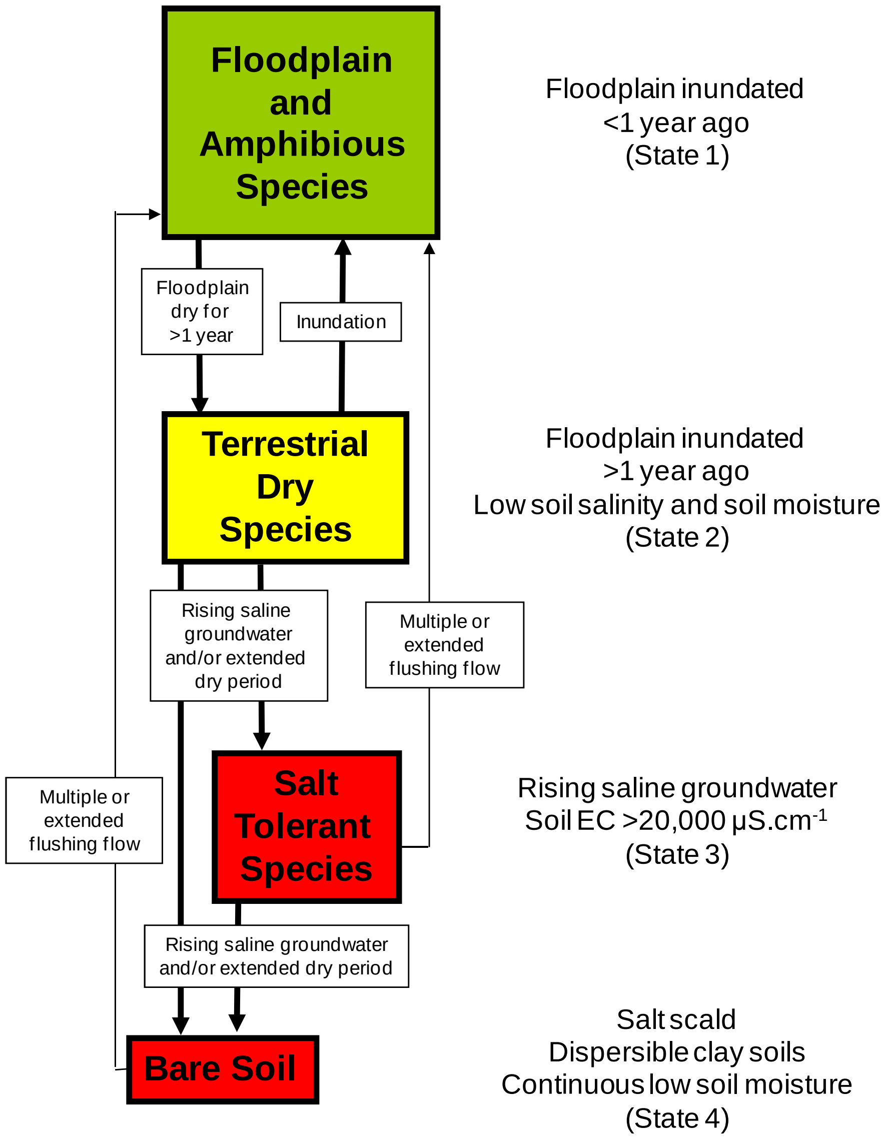

Figure 3.5 Conceptual model of floodplain plant community dynamics showing states based on functional groups present and the major factors that result in changes between states. Source: Nicol *et al.* 2010a.

Results from Chowilla (Gehrig *et al.* 2012b) and Pike (Holland *et al.* 2013) generally supported the model except the transition from states three or four to state one. Results from the 2005 in–channel flow pulse suggested that once an area had become salinised it would take a large amount of water and that it would be difficult to return to state one (Zampatti *et al.* 2011). However, monitoring results from Chowilla (Gehrig *et al.* 2012a) and Pike (Holland *et al.* 2013) showed that badly salinised areas also retained the capacity to respond. However, it is unknown whether the source of propagules was from the local sediment seed bank or dispersal into the area. Gehrig *et al.* (2012a) reported that species richness was much higher in quadrats that sampled a strand line, which provides evidence that hydrochory may be important for species persistence.

In contrast to the floodplain, the aquatic plant community was not monitored extensively; however, baseline surveys and fish habitat assessments were undertaken between 2004 and 2012. During 2007 and 2008 river flows were very low and there was a decrease in the turbidity of River Murray. During this period extensive beds of the submergent native species *Hyrilla verticillata* and exotic submergent species *Elodea canadensis*, established in the River Murray Main Channel and numerous wetlands (Marsland *et al.* 2010). The beds were so dense in the Lock 2 weir pool that they posed a risk to swimmers. Other submergent species such as *Potamogeton crispus, Zanichellia palustris* and *Vallisneria australis* were also abundant upstream of Lock 1 during the period of extreme low flow between 2006 and 2010 (Nicol *et al.* 2006; Marsland and Nicol 2007; Marsland and Nicol 2008a; Zampatti *et al.* 2011c). Since the 2010–11 floods, there has been a decrease in submergent species throughout the Riverine Corridor (except in the Lower Lakes) (Gehrig *et al.* 2012a; J. Nicol unpublished data). It is likely that the increased flow and water level restricted the euphotic zone depth, or plants were physically uprooted and transported downstream.

Role of aquatic and floodplain understorey vegetation in riverine function

The role of floodplain and aquatic vegetation in ecosystem function is not well understood in the lower River Murray. Aquatic plants have been identified as important primary producers (e.g. Blindow *et al.* 2006; Mnaya *et al.* 2007; Silva *et al.* 2009; Noges *et al.* 2010), provide physical habitat for fish (e.g. Balcombe and Closs 2004; Katende 2004; Wedderburn *et al.* 2007), birds (e.g. Kapa and Clarkson 2009; Blackwood *et al.* 2010) and invertebrates (e.g. Papas 2007; Walker *et al.* 2013; Yozzo and Osgood 2013), improve water quality (e.g. Maddison *et al.* 2009; Konnerup and Brix 2010; Zhu *et al.* 2010; Vymazal 2011) and control erosion (Williamson *et al.* 1992; Caffrey and Beglin 1996; Abernethy and Rutherfurd 1998). Significant positive interactions between fish and plant species were observed in the lower River Murray between Wellington and the New South Wales border (Marsland *et al.* 2010, Zampatti *et al.* 2011c, Beyer *et al.* 2011; Leigh *et al.* 2012). For example, golden perch were significantly associated with the emergent plant *Phragmites australis,* while carp were associated with the submergent species *Vallisneria australis*. Whilst there are data regarding the associations between fish and aquatic macrophytes for the lower River Murray upstream of Wellington, this information is not available for the Lower Lakes and Coorong. Similarly, the habitat value of aquatic and floodplain understorey plants for reptiles, mammals, frogs and woodland birds is less well understood.

The contribution of aquatic and floodplain plants to the food web of the lower River Murray is also not well understood and often based on work undertaken in tropical or temperate systems with regular flooding cycles (e.g. Junk *et al.* 1989). Most plant species present on the floodplains of the lower River Murray (and other Australian arid zone streams) recruit as water levels recede to take advantage of the bare substrate with high soil moisture (Capon 2003; Nicol 2004; Capon and Brock 2006; Capon 2007; Gehrig *et al.* 2012; Holland *et al.* 2013). This recruitment of floodplain plants means that the lower River Murray floodplain is probably several orders of magnitude more productive than the surrounding arid landscape shortly after floodwaters have receded. However, it is unlikely that this pulse of productivity is a large contributor to the aquatic food web because there is no direct pathway to the river for the carbon fixed by the understorey vegetation before the next flood (which is usually after plants have completed their life cycle). It is likely that the understorey floodplain vegetation makes a larger contribution to the terrestrial food web than to the riverine food web, which may have been an important ecosystem service provided by the floodplain that has been reduced due to reduced flooding frequencies.

Impacts of the proposed basin plan on aquatic and floodplain understorey vegetation

A detailed assessment of the impacts of the 2,750 GL yr^{-1} and 2,800 GL yr^{-1} scenarios in the proposed basin plan is presented in Lamontagne *et al.* (2012). Key points of the assessment of the proposed basin plan regarding aquatic and floodplain understorey vegetation are (from Lamontagne *et al.* 2012):

- The proposed basin plan will provide increased recruitment opportunities for floodplain understorey vegetation in temporary wetlands and low elevation floodplains (areas inundated with flows up to 80 GL day^{-1}) compared with the current regime due to increased medium sized floods and greater water level variability.

- High elevation floodplains and temporary wetlands (areas inundated with flows >80 GL day^{-1}) will require natural overbank flows to be inundated, which will not be managed under the proposed basin plan. Hence, there will be no improvement for floodplain understorey vegetation in these areas. It is likely that these areas will remain dominated by terrestrial or salt tolerant species under the proposed basin plan.
- There will be an increase in the distribution of littoral emergent species due to the increase in the littoral zone because of increased water level variability.
- Submergent species will probably not benefit due to increased periods of deeper water in wetlands; however, the improved littoral zone may result in better water quality that will benefit submergent species.
- Extreme low–water levels and elevated salinities may still occur infrequently in the Lower Lakes under extended drought conditions under the proposed Basin Plan; hence there is still potential for fringing wetlands to dry downstream of Lock 1 albeit with lower frequency than with current conditions. Shoreline (Lower Lakes) and floodplain (River Murray between Wellington and Lock 1) wetlands provide almost the entire habitat for submergent and emergent species hence these species may still decline at times under the proposed basin plan.
- It is unlikely that salinities in the North lagoon will support Ruppia megacarpa under the proposed basin plan; however, lower salinities and higher water levels in the South Lagoon may result in an increase in the distribution and abundance of Ruppia tuberosa.

The water available for the environment in the Basin Plan was increased to 3,200 GL yr-1; however, the ecological benefits were not assessed to the same extent as with the initial proposed Basin Plan. It is likely that the ecological benefits of returning 3,200 GL yr^{-1} to the river will be similar to the original proposed Basin Plan with the extra water increasing the magnitude and duration of flooding (providing water delivery constraints are relaxed). This additional water will further reduce the chances of low water levels and high salinities in the Lower Lakes, and reduce salinities in the Coorong.

3.3. Birds and mammals

The River Murray and its floodplain provide internationally significant habitat for waterbirds and also support a wide range of terrestrial birds, including rare and threatened species. Many of these bird species have suffered regional and national population declines in the last 25 years (Kingsford *et al.* 1999a; Ford *et al.* 2001; Porter *et al.* 2006). Floodplain environments and the productivity supported by flooding are also important for native mammals and reptiles; however the responses of these fauna groups to floodplain management are far less well studied. In general, responses of floodplain mammals, terrestrial birds, and reptiles to flooding have not been well documented in Australia to–date; and there have been few inter–linked and multi–disciplinary studies of primary and secondary productivity responses to flooding. Consequently, predicting the effects of managed floods and changes in flood regimes upon floodplain fauna has been difficult.

There are several features of the flood regime that are important for waterbirds, including flood frequency, duration, depth, rate of fall, and the duration of the period between floods (Rogers 2011 and references therein). These features affect both the availability of suitable habitat and the availability of food resources (Kingsford and Norman 2002; Kingsford *et al.* 1999b). Flooding stimulates breeding in most waterbirds; recent work has found that for colonial breeding large waders there are clear thresholds of flow producing a >0.7 probability of attempted breeding when a daily flow threshold was exceeded for 30–50 days (Arthur *et al.* 2012). However flood requirements vary and are plastic within and among species, so environmental flow planning needs to take into account the needs of a wide range of species, aiming to maximise breeding, recruitment and between–flood survival for maintenance of both diversity and abundance (Rogers 2011).

Floodplains and riparian zones are host to a wide range of woodland bird species (Fisher and Goldney 1997; Major *et al.* 2001; Antos and Bennett 2005). These include resident, nomadic, and migratory groups, and many threatened and declining species (Reid and Fleming 1992; Reid 1999). Some are dependent upon floodplain vegetation communities, while others occupy floodplains but are also resident in other woodland types (McGinness *et al.* 2010). Floodplain woodlands and forests are thought to be important refuges for woodland bird populations because a) floodplain woodlands and forests comprise some of the largest and most continuous vegetation remnants in south–east Australia; and b) floods intermittently supply water, sediment and nutrients that drive greater primary and secondary productivity than found in woodlands not subject to flooding (McGinness *et al.* 2010). Changes in flooding are likely to influence woodland bird populations directly through availability and quality of open water, and indirectly via changes in aquatic and terrestrial primary and secondary productivity, vegetation condition and structure, and vegetation distribution (Kingsford 2000; Horner *et al.* 2009). McGinness *et al.* (2010) suggested that floodplain communities and their requisite floods are of great importance for the persistence, productivity and diversity of woodland birds in Australian drylands.

Bird community composition and bird abundance differ among floodplain vegetation communities, and may even differ in response to flood frequency (McGinness *et al.* unpublished data). This has significant implications in the context of reduced flooding or abandonment of higher–elevation communities and transitions from one vegetation community to another. Higher–elevation *E. largiflorens* woodland communities support greater abundances of birds, in particular greater densities and a wider variety of ground–foraging and hollow nesting species, than other vegetation types including more frequently flooded *E. camaldulensis* (River Red Gum; Carpenter 1990; Antos and Bennett 2005). *Eucalyptus largiflorens* communities maintain high bird species richness in both breeding and non–breeding seasons, while *E. camaldulensis* communities tend to be dominated by nomadic or migratory species during the spring–summer breeding season (Antos and Bennett 2005). Carpenter (1990) attributed the relative persistence, diversity and abundance of birds in the *E. largiflorens* community to the fact that this floodplain habitat shares features of both riverine and arid vegetation types. It is also likely that the intermediate nature of the flood regime in this community is important, with insufficient flooding occurring to exclude possibly flood–sensitive species but enough flooding to encourage species that benefit from inundation and its effects on vegetation productivity (McGinness *et al.* 2010).

Many mammal species present in the River Murray and its floodplains in the past are now rare. The only abundant and widespread native mammals are bats, kangaroos, and the brush–tailed possum (*Trichosurus vulpecula*). Currently the greatest diversity of mammals appears to be in river red gum forests, which contain several species of bats, as well as yellow–footed antechinus (*Antechinus flavipes*), water rats (*Hydromys chrysogaster*), black wallaby (*Wallabia bicolour*), sugar glider (*Petaurus breviceps*) and platypus (*Ornithorhynchus anatinus*). Black box woodland also supports bats, kangaroos, and brush–tailed possums, and the presence of these species is dependent on specific landscape and woodland characteristics, for example landscape complexity, shrub and log cover, presence of hollow–bearing trees and woodland patch size (Lewis 2006). Little is known of the extent to which native mammals in floodplains rely on the productivity produced by flooding, however floodplain river energy sources have been quantitatively shown to support kangaroos (Iles *et al.* 2010). The effects of changes in flooding regime upon mammals are also generally poorly understood.

3.4. Frogs

There are eleven frog species along the River Murray corridor in South Australia. Each frog species has different habitat associations, water regime preferences, capacity to tolerate drying and temperature ranges (Table 3.4). Of the eleven species present, the southern bell frog (*Litoria raniformis*) is the only nationally threatened species (Vulnerable listing under *EPBC Act* 1999) although several other species have State or regional listings. Frogs and tadpoles occur in wetland vegetation, which provides habitat resources such as food, sites for egg laying and cover from predators such as fish, snakes and birds. Male frogs will typically call whilst floating on algal or plant mats in the open water, from vegetation near the edge of the wetland or from depressions or ditches near a wetland (refer Anstis 2007). For example, the southern bell frog males will call from recently inundated riparian vegetation (August to January, and often adjacent to a permanent refuge) and have extremely flexible tadpole maturation periods from 2.5 to15 months (Wassens 2005). By contrast, the highly opportunistic *Crinia* spp. will rapidly disperse following rain at any time of the year to make use of temporary habitats that may only be wet for six weeks, which is sufficient for tadpoles to mature (Anstis 2007). Temperature and food availability have also been show to affect tadpole development, with tadpoles grown at higher temperatures with high food availability metamorphosing earlier and being larger (Newman 1998).

Table 3.4 Vital attributes of frogs. Adapted from Anstis 2007; Gonzalez *et al.* 2011; Turner *et al.* 2011; SAMDB NRMB unpub. data; K. Mason and R. Turner, pers. comm. Qualitative terms (e.g. short, long) are those of the original authors.

Species	**Habitat associations**	**Recruitment needs**
Southern Bell Frog ***(Litoria raniformis)*** **EPBC–listed**	Variable water regime (ephemeral or temporary for breeding); use permanent wetlands as refuge; associated with complex wetland vegetation and low salinity $<500~\mu\text{Scm}^{-1}$ Prefer lignum, river red gum or black box, diverse emergent plants and herbs, floating and submerged plants and inundated grasses;	Recently inundated vegetation; Males call spring to autumn from vegetation; Metamorphosis summer to autumn, 2.5 to 15 months; Tadpoles present in November, absent from February onwards. Highly mobile, moving up to 2 km between wetlands, especially during floods
Long–thumbed frog ***(Limnodynastes fletcheri)***	Range of natural and built aquatic habitats; prefers seasonally inundated wetlands; wet ≥ 6 months.	Temporary, shallow, well–vegetated wetlands; Males call spring to autumn, and during mild winters from vegetation (after heavy rains); Opportunistic breeders with metamorphosis occurring anytime (short maturation).
Murray Valley froglet ***(Crinia parinsignifera)***	Dominant in SA Riverland areas; habitat generalists; prefer abundant aquatic vegetation or submerged terrestrial vegetation; Desiccation avoidance poorly understood.	Breed opportunistically; exploit highly ephemeral wetlands (flood and rain–fed) Dispersal poorly understood; Males call most of the year from the ground and grasses; Tadpole maturity short, absent by November
Common froglet (***Crinia signifera)***	Dominant below Lock 1; habitat generalists; prefer abundant aquatic vegetation or submerged terrestrial vegetation; Desiccation avoidance poorly understood.	Breed opportunistically; dispersal poorly understood; Males call most of the year; Tadpole maturity 6 weeks to 3 months or more, absent by Nov.; Tadpole salinity tolerance up to 9,400 μScm^{-1} EC.
Southern brown tree frog ***(Litoria ewingii)***	Habitat generalist; temporary and permanent wetlands (dense reeds); terrestrial and built habitats; highly mobile; more common downstream of Walker Flat.	Exploit highly ephemeral wetlands (flood and rain–fed); Males call after rain with peak breeding in early spring and autumn Tadpole maturity short
Spotted grass frog ***(Limnodynastes tasmaniensis)***	Habitat generalist; readily colonise new wet areas; very resilient species	Breed opportunistically; highly dispersive; Tadpoles generally more abundant in aquatic vegetation; Males call spring, summer, autumn and mild winters especially after rain Tadpole maturity short (at least 3 months), Tadpole salinity tolerance $< 6000~\mu\text{S/cm}$ EC
Eastern banjo frog (L***imnodynastes dumerili)***	Very wide distribution from coast to inland; often associated with slopes and ranges.	Rainfall dependent; range of water bodies; Males migrate long distances; will breed in permanent wetlands in any season; highly fecund; Males call most intensely after rain, in cooler months. Tadpole maturity 5–6 months (spring to autumn).
Peron's tree frog (***Litoria peronii)***	Range of habitats; shelters in tree hollows and bark in dense river red gum stands; prefers trees and dense reeds; known to exist in terrestrial habitats	Males call late–spring/summer; Can breed in permanent, deep, open water; rarely breeds in very shallow well vegetated habitats; optimal temporary floodplain reaches; Tadpole maturity at least 3.5 months, low salinity tolerance
Burrowing frog/Painted Frog ***(Neobatrachus pictus)***	Not dependent on river; wide range of arid and semi–arid areas; aestivate and form a cocoon to avoid desiccation; soils suitable for burrowing	Temporarily inundated sites; flooded and rain–fed wetlands; not highly dispersive; dispersal depends on rainfall; Males call autumn and winter after rain.
Sudell's frog ***(Neobatrachus sudellii)***	Not dependent on river; wide range of arid and semi–arid areas; aestivate and form a cocoon to avoid desiccation; soils suitable for burrowing	Temporarily inundated sites; flooded and rain–fed wetlands; not highly dispersive Males call after rain. Tadpoles can overwinter after autumn breeding; metamorphose in spring / early summer.
Brown Toadlet/Bibron's Toadlet (***Pseudophryne bibronii***)	Not dependant on river	Males call February to June, particularly after heavy rain; usually near wetlands or creeks; Tadpole maturation 120–180 days, metamorphosing late winter to summer.

Environmental conditions and the length of the wet phase during tadpole development are critical for successful recruitment. If water does not persist until the tadpoles have metamorphosed into frogs, they may be trapped and desiccated or heavily preyed upon as the aquatic habitat reduces. For example, calling southern bell frog males were observed at Clayton Bay in 2009 in wetlands with salinities >10,000 μScm^{-1} EC, however, no tadpoles were captured. This is consistent with southern bell frog tadpoles having only been detected in low salinity water (<600 μScm^{-1} EC; SAMDB NRMB unpub. data). Other species may have greater environmental tolerance or adaptability such as *Limnodynastes* sp. and *Crinia* sp. tadpoles, which were found in up to 9,300 μScm^{-1} EC in Lakes Alexandrina and Albert during the recent low flows (SAMDB NRMB unpub. data). Poor water quality, particularly hypoxia has been shown to lead to deformities, death of embryos and early hatching. Hatching at early stages of development is likely to have a negative impact on growth, ability to avoid predation and reproductive success of adults (Mills and Barnhart, 1999; Seymour *et al.*, 2000).

Most of the other frog species are less sensitive than the southern bell frog to the frequency, timing, extent and duration of water regime. Thus by managing wetlands to support the vulnerable southern bell frog, primarily through providing complex vegetation across most of the elevation gradient and regular flooding, suitable habitat for other native frog species is likely to be provided. For example, long–thumbed frogs have similar habitat requirements and provided that the seasonal inundation period was extended from August and into March other species could also complete their life cycles. Sites in the lower River Murray support this hypothesis, as all sites where Southern bell frogs were calling also had a high diversity of other frogs (K. Mason pers. comm.).

Unless the frog has the capacity to burrow (e.g. burrowing frog, Sudell's frog) they are dependent on permanent aquatic or moist environments for refuge. Eastern banjo frogs can burrow to escape desiccating conditions (Tyler 1994) and it is thought that they are likely to burrow deeper to find moist soil as the water table drops. In contrast, the long–thumbed frog has limited tolerance for wetland drying and is lost if there are no permanent refuge sites nearby (Wassens 2011). Maintaining permanent refuges within 1 km will most likely increase resilience and local persistence. Conversely, eastern banjo frogs emerge to forage during heavy rain, which may extend the length of time individuals can persist between floods (Wassens 2011). However, these frogs are sensitive to desiccation and their distribution may be restricted by soil type, which means that eastern banjo frogs may need to retreat to permanent refugia and rely on floods to re–colonise wetlands upon re–wetting. Peron's tree frog and spotted grass frog are probably the species most resilient to extended drying, being found at isolated wetlands when re–wet after extended dry periods (see Gonzalez *et al.* 2011 for review). Although there are no direct benefits of drying to frogs there may be indirect benefits such as increased diversity and abundance of aquatic vegetation and extirpation of predatory fish.

Wetting and drying at a wetland scale will maintain isolated frog populations, however weir pool raising or floods are required to inundate riparian vegetation to support dispersal and optimise recruitment and population connectivity. Wetlands that are refilled by pumping and have a prolific aquatic vegetation response will provide particularly good habitat for frogs because of the abundance of food and the limited predation and competition by fish (NB: small predatory fish such as eastern gambusia and carp gudgeons may pass through pumps, Hillyard pers. comm.). For example, pumping water into Markaranka wetlands in 2006 and 2009 resulted in low fish abundance, a diverse range of inundated vegetation (including lignum) and an abundant and diverse frog response (K. Mason pers. comm.). Similarly, weir pool raising or flooding that inundates depressions behind the main wetland basin may also support abundant frog populations (e.g. Sweeney's Lagoon in 2006, K. Mason pers. comm.).

3.5. Fish

The flow regime is the overarching driver of riverine ecosystem structure and function (Poff and Allan 1995; Sparks *et al.* 1998), influencing the distribution and abundance of all aquatic biota, including fish. Variability in the flow regime drives fish assemblage structure both directly, by influencing critical life history processes including spawning, migration and recruitment (Humphries *et al.* 1999; Junk *et al.* 1989; King *et al.* 2009; Welcomme 1985), and indirectly, by influencing floodplain inundation, productivity, channel morphology, hydraulic conditions, the distribution of aquatic vegetation and diversity of structural elements, and subsequently habitat availability and selectivity (Nestler *et al.* 2012). A fish response model, developed by Ye *et al.* (2009), is used to illustrate the life cycle of a generalised fish species and the relationship between critical life history processes (Figure 3.6) and the habitat / flow environment provided by a given set of river conditions (Figure 3.7). Variability in the flow regime influences biotic communities throughout the river continuum from the source of a river through to its estuary. Nonetheless, the importance of different aspects of the flow regime and how components are manifest may vary between different aquatic environments (i.e. floodplain river reaches, lakes and estuaries). This premise holds true for the Coorong, Lower Lakes and Murray Mouth (CLLMM) region of the lower River Murray and as such the main river channel, anabranch systems and floodplains above Wellington will be discussed separately to the CLLMM region.

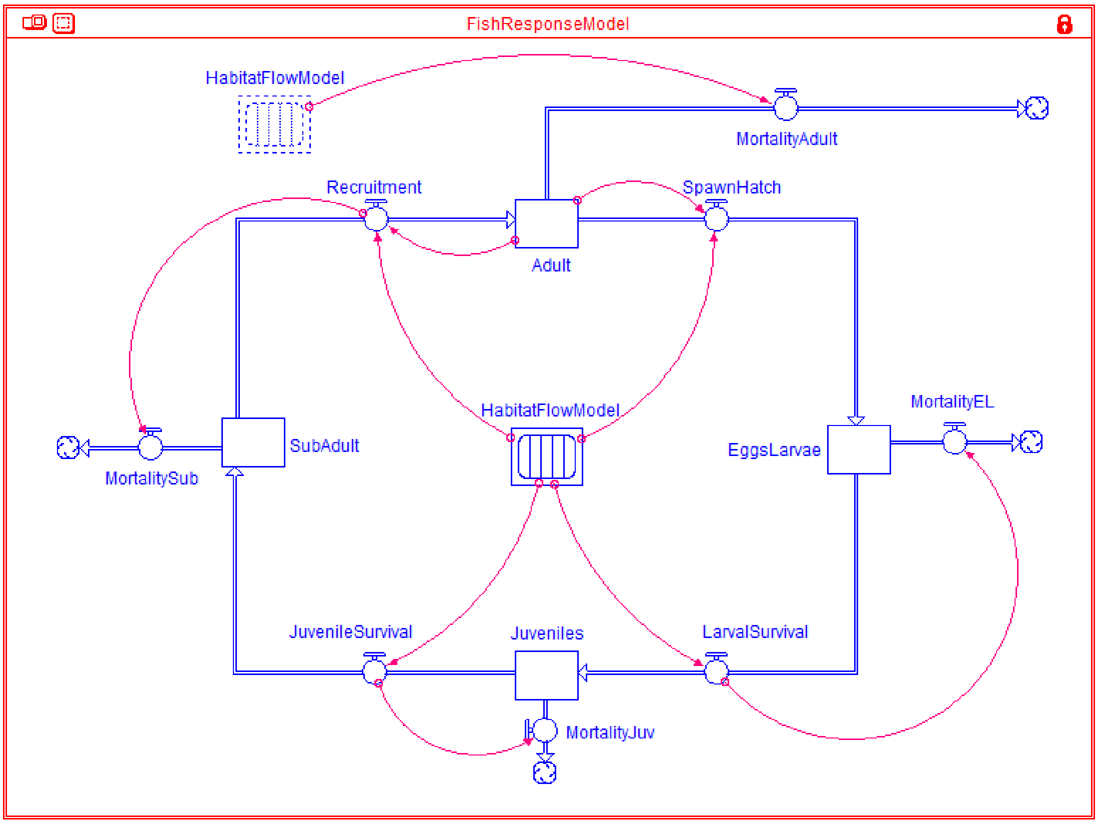

Figure 3.6 Fish Response Model illustrates the importance of understanding the complex interplay between physical habitat and flow regime in population dynamics of selected fish species within the river system. Source: Ye *et al.* 2009.

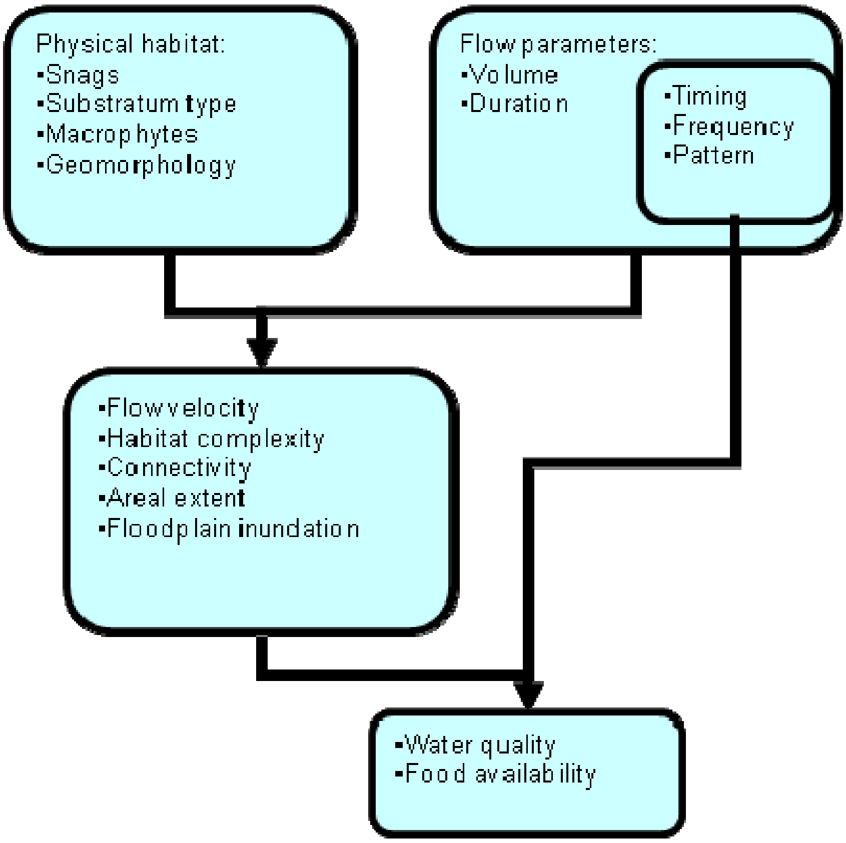

Figure 3.7 Habitat Flow Model illustrating the interaction between flow and physical habitat. Source: Ye *et al.* 2009.

In Australia, a number of studies have been undertaken to improve understanding of the significance of flows and/or floodplain inundation on key aspects of fish population dynamics (life history processes), particularly spawning and recruitment (e.g. Cheshire and Ye 2008a; Cheshire and Ye 2008b; Gilligan and Schiller 2003; Harris and Gehrke 1994a; Humphries *et al.* 1999a; Humphries and Lake 2000; Humphries *et al.* 2002; King *et al.* 2003; King *et al.* 2009; King *et al.* 2010; Ye *et al.* 2008; Balcombe *et al.* 2007; Balcombe and Arthington 2009; Zampatti and Leigh 2013). Indeed, two analogous fish recruitment models have been developed based on observations from temperate Australian floodplain rivers, the *flood recruitment model* (FRM) (Harris and Gehrke 1994b) and the *low flow recruitment hypothesis* (LFRH) (Humphries *et al.* 1999). The FRM proposes that a rising hydrograph is a cue for spawning in some species, and that floodplain inundation provides both an increase in productivity and suitable habitat for fish larvae and early juveniles. Conversely, the LFRH proposes that the falling–limb and low–flow components of the hydrograph provide the greatest opportunity for recruitment success, particularly for small–bodied species. Recent research suggests that both models have merit and are applicable to different species within the lower River Murray.

In the last decade, flow conditions within the lower River Murray have been such that there were opportunities to investigate the influence of flow variability (i.e. low flow, within–channel flows, overbank flows) on different life history processes (i.e. spawning, recruitment and migration) of different species (Cheshire and Ye 2008a, b; Leigh *et al.* 2008; Ye *et al.* 2008; Bucater *et al.* 2009; Leigh and Zampatti 2011; Cheshire *et al.* 2012; SARDI unpublished data). Findings from investigations of spawning and recruitment are discussed in terms of groups of species relating to 'spawning modes' as described by Humphries *et al.* (1999) (i.e. Mode, 1, 2, 3a and 3b spawners), which differ based upon the influence of flow regime (direct or indirect) on spawning. Mode 1 species (e.g. Murray cod) spawn around the same time every year, with spawning initiated through a combination of circa–annual rhythms and temperature, they do not require an increase in water discharge to trigger spawning (Humphries 2005; Koehn and Harrington 2005; Koehn and Harrington 2006; Humphries *et al.* 1999b). Mode 2 species (e.g. golden perch) are flood/flow cued spawners, requiring increases in discharge rates to initiate spawning and promote larval survivorship (Gilligan and Schiller 2004; King *et al.* 2005; Mallen–Cooper and Stuart 2003; Ye 2005; Humphries *et al.* 1999; Ye *et al.* 2008). Mode 3a (e.g. flat–headed gudgeon) and 3b (e.g. Murray rainbowfish) species will spawn and recruit under a range of flow conditions, although none are believed to be cued by changes in flow regime, many species respond well to low flow conditions, following the low flow recruitment hypothesis (Humphries *et al.* 1999).

Results from investigations of fish spawning and recruitment dynamics in both anabranch and main channel environments in the South Australian River Murray support Humphries *et al.* (1999) proposed groupings of species by modes. Murray cod, a mode 1 species, has been shown to spawn annually, at specific times, irrespective of increases in flow (Bucater *et al.* 2009; Cheshire and Ye 2008a; Cheshire and Ye 2008b; Leigh *et al.* 2008) Cheshire *et al.*2012). Nonetheless, it is believed that increases in flow and flooding benefit larval survivorship and subsequent recruitment, and strong year classes have previously been associated with years of higher flows and flooding in SA (Ye *et al.* 2000; Ye and Zampatti 2007). Furthermore, spawning and recruitment of golden perch, a mode 2 species, has been detected during both within–channel and overbank flows, interspersed with years of no spawning and recruitment during low flow years (Ye 2005; Ye *et al.* 2008; Cheshire *et al.* 2012). Importantly, enhanced recruitment observed in years with increased within–channel flow were notable when there was a lack of floodplain inundation, suggesting within–channel flow pulses can elicit positive responses from this species (see Mallen–Cooper and Stuart 2003). Humphries *et al.* (2002) classify the non-native common carp as a mode 2 species, but spawning and recruitment in the lower River Murray typically occur annually (Smith and Walker 2004). Nonetheless, enhanced spawning and recruitment occurs with floodplain inundation (King *et al.* 2003; Stuart and Jones 2006) and recent evidence suggests that enhanced spawning and recruitment may occur in the lower River Murray with naturally elevated flows (Cheshire *et al.* 2012; SARDI unpublished data) and managed local rises in water levels (Bice and Zampatti 2011). The remaining species, typically small–bodied fish, fall within modes 3a and 3b, and spawning and recruitment is typically detected annually in the main channel of the lower River Murray, although there is some evidence to suggest that recruitment is greatest during times of low flow. Nevertheless, a recent study in the lower River Murray during the protracted drought suggests that prolonged low flow conditions may, in time, have a negative impact on previously defined low flow spawners (Bucater *et al.* 2009). Additionally, Wedderburn *et al.* (2013) observed that increased zooplankton abundance corresponded with enhanced recruitment of Murray hardyhead (listed as *endangered* under the *Environment Protection and Biodiversity Conservation Act 1999*) following inundation of a previously dry irrigation channel, and suggested flooding may facilitate early life stage survival in this species.

High river flows and flooding in 2010/11 allowed investigation of the *flood recruitment model* (FRM) (Harris and Gehrke 1994b) in the lower River Murray. Indeed, spawning and recruitment in some species, including golden perch appeared enhanced following flooding (Cheshire *et al.* 2012; Zampatti and Leigh 2013), potentially due to increased productivity following floodplain inundation and enhanced lateral connectivity. High flows and flooding also resulted in dramatic changes to habitat (hydraulic and physical) character and availability within the lower Murray. In concurrence with the habitat flow model proposed by Ye *et al.* 2009 (Figure 3.7), high flows and accompanying increased flow velocities and turbulence resulted in changes to instream vegetation communities, including a general loss of submerged macrophytes and inundation of the floodplain. Concurrently, there were significant changes in the fish assemblage, including marked declines and absences of small–bodied fish species in main channel habitats (SARDI unpublished data). Use of inundated floodplain as habitat by several species was also noted during this period (SARDI unpublished data).

Variability in the flow regime of the lower River Murray is also important in regards to influencing fish life–history processes and assemblage structure due to its effect on longitudinal and lateral connectivity, and thus fish movement and migration. Several species may undertake long distance upstream migrations (e.g. golden perch, Murray cod, common carp) and flow pulses and floods provide both cues and connectivity to enable these migrations. Recent radio–tracking work investigating the movement of Murray cod in the Chowilla anabranch system noted long distance upstream movements (>50 km) of several individuals during high flow conditions (SARDI unpublished data). Large–scale movements of Murray cod in response to high flows have also been reported in other areas of the Murray–Darling Basin (Koehn and Nicol 1998). Such movements are likely undertaken for the purpose of spawning and are likely important for maintaining metapopulation dynamics across large spatial scales. Smaller scale movements both longitudinally and laterally, between the main channel and off-channel habitats, have been observed in small-bodied native species (Baumgartner *et al.* 2008; Lyon *et al.* 2010; Connalin *et al.* 2011). Whilst the purpose of such movements is unclear, they are undoubtedly important, potentially occurring in relation to dispersal, spawning and the exploitation of newly available (inundated) habitat.

Variability in the flow regime is equally important in structuring fish assemblages in the Lower Lakes and Coorong, as it is in the main river channel, however, the mechanisms that drive population and assemblage dynamics may be different. Fish assemblages within the Lower Lakes and Coorong are undoubtedly influenced by upstream processes including the downstream transport of sediments and nutrients but processes occurring locally within these environments may differ. In the main river channel, greater emphasis is placed upon hydraulic diversity, provision of cues for critical life history processes, longitudinal and lateral connectivity and floodplain/wetland inundation, but fish communities of the Lower Lakes are more influenced by water level variability than floodplain inundation, and connectivity between the Lower Lakes and Coorong. Water level variability within Lakes Alexandrina and Albert, dictated by a combination of variable inflows, evaporation, wind seiche and barrage releases, is believed to be primarily responsible for patterns of vegetation diversity and distribution, and hence, habitat availability and quality for many fish species. Low inflows, drought and resulting water level recession in the Lower Lakes over the period 2006 – 2009, and accompanying elevated salinity, habitat fragmentation and loss of submerged aquatic macrophytes, likely resulted in the extirpation of Yarra pygmy perch (*Nannoperca obscura*) and declines in southern pygmy perch (*Nannoperca australis*) and Murray hardyhead (*Craterocephalus fluviatilis*) in the wild (Bice *et al.* 2011; Wedderburn *et al.* 2012). Additionally, over this period generalist freshwater fish species with broad tolerance limits and euryhaline estuarine fish species came to dominate the assemblage (Wedderburn *et al.* 2012). Water level fluctuation within a natural range thus must be maintained through inflows to maintain habitat connectivity, and favourable water quality and habitat.

Hydrological connectivity as dictated by flow has a marked effect on the fish community of the Coorong, Lower Lakes and Murray Mouth (CLLMM) region. Several diadromous fish species, those requiring migration between freshwater and estuarine/marine environments to complete their lifecycles, are present within the region. Loss of flow to the Coorong and subsequent loss of connectivity in the past has resulted in greatly diminished recruitment and abundance of diadromous species in the region due to obstruction of spawning migrations, loss of conditions suitable for recruitment and loss of migratory cues (Zampatti *et al.* 2010, 2011b). Conversely, when connectivity is facilitated, diadromous species are typically common and exhibit regular seasonal recruitment (Zampatti *et al.* 2011a).

Freshwater inflows to the Coorong not only influence connectivity and subsequent fish migration, but also have a marked influence on the physico–chemical environment of the Coorong and in turn the resident fish community. Spatio–temporal variability in salinity is widely regarded as a primary driver of estuarine fish species distribution and assemblage structure (Barletta *et al.* 2005), and research in the Coorong over the past seven years has reinforced this paradigm. During drought from 2007 – 2009, freshwater inflows to the Coorong diminished and salinities increased resulting in fish assemblages which were increasingly characterised by marine species (Ye *et al.* 2012a; Zampatti *et al.* 2010). Estuarine fish habitat and recruitment of a number of estuarine and diadromous species were also negatively impacted (Noell *et al.* 2009; Zampatti *et al.* 2010). Nonetheless, fish assemblages were characterised by freshwater species following significant inflows in 2010 (Ye *et al.* 2012a; Zampatti *et al.* 2011a), and shifted to estuarine dominant assemblages following continued flow in 2011 (Ye *et al.* 2012b). Enhanced recruitment was detected for several species including smallmouthed hardyhead, mulloway and congolli (Ye *et al.* 2012b). Additionally, spatio–temporal variability in salinity was found to significantly influence the distributional range of estuarine species within the Coorong, with broad–scale reductions in salinity over the period 2010–12, resulting in greatly increased ranges for several fish species including black bream, greenback flounder, yelloweye mullet, sandy sprat, small mouthed hardyhead and Tamar goby (Ye *et al.* 2011; Ye *et al.* 2012b). In summary, estuaries are highly dynamic environments and are inherently dependent upon riverine flow variability to maintain this dynamism.

3.6. Micro- and macro-invertebrates

Aquatic macro- and micro-invertebrates are a large, heterogeneous group of organisms that includes crustaceans, insects, worms, rotifers, leaches, molluscs, decapods and sponges (Brookes et al. 2009). Incorporating herbivores, detritivores and predators, macro– and micro–invertebrates are important consumers within wetland ecosystems, incorporating a wide range of functional groups (*e.g.* filtering collectors, predators, scrapers). They also provide a vital pathway in trophic cascades between bacterial and algal production and the higher order trophic levels (Bunn et al. 2003). By consuming bacteria and microalgae, they indirectly alter nutrient cycles, transfer energy to secondary consumers including macro-invertebrates, fish and waterbirds (Jenkins and Boulton 2003). Many waterbirds feed extensively on micro-invertebrates (Crome 1985). Invertebrates are a key food resource for breeding waterfowl as they provide the protein source required for egg and nestling development. The responses of waterbird guilds that are piscivorous, herbivorous, reliant on aquatic macro–invertebrates 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). The increase in habitat and food resources (the productivity boom; Bunn *et al.* 2006) associated with floods is considered to provide a direct benefit for native fish (Harris and Gehrke 1994a, Bunn and Arthington 2002, Arthington et al. 2005), many of which may rely on micro-invertebrates for their first feed (see Jenkins and Boulton 2003).

Invertebrate community composition is influenced by many factors including water and habitat quality and quantity, linkages between wetland and riverine environments and trophic interactions (Reid and Brooks 2000). Increasing (i) habitat complexity, (ii) protection or cover from predation and (iii) diversity of food sources for a range of many functional feeding groups, may enable a variety of different functional feeding groups to co–occur (Boulton and Brock 1999). Many invertebrate taxa are known to be sensitive to water quality and habitat disturbance. Because of this sensitivity and their often rapid response to environmental change, invertebrates are frequently used as an indicator of river health and environmental disturbance (Reid and Brooks 2000, Jenkins et al. 2005).

Role of hydrology

At the basin scale, the heterogeneity of habitats is considered to be responsible for the wide diversity of the micro- and macro-invertebrate community of the MDB (Hillman and Quinn 2002). Hydrology is also a major determinant of invertebrate community composition through its influence on habitat availability and structure (*e.g.* macrophytes) (Puckridge et al. 1997, Brauns et al. 2008). In Australia, extreme temporal flow variability (Puckridge et al. 1998) and episodic lateral connectivity with broad floodplains is believed to stimulate episodic productivity ‘booms’ (McComb and Qiu 1998). Inundation of temporary wetlands creates conditions substantially more productive than those found in permanent wetlands.

Flow regulation promotes the abundance of some habitat types at the expense of others. For example, in the lower River Murray, it is considered that the high abundance of submerged and emergent vegetation that currently exists is an artefact of river regulation, as photographs taken prior to construction of the locks and weirs indicate that vegetation within the channel was sparse (see Sheldon and Walker 1998). The proliferation of vegetation within the channel may provide some substitution for the decline in floodplain wetlands. However, this will have altered the temporal and spatial distribution of invertebrates (Sheldon and Walker 1997).

In some reaches, construction and operation of in-stream structures and modification of the flow regime may decrease habitat fragmentation by creating permanent connection between macrohabitats (e.g. river channel, anabranch, wetland), that would otherwise only intermittently be connected. In other reaches, regulation may increase fragmentation by decreasing the frequency at which floodplain habitats are inundated (Sheldon and Thoms 2006). Sheldon and Walker (1998) demonstrate the importance of macrohabitats, mesohabitat (pool, riffle) and microhabitats (snag, aquatic macrophytes) in shaping macroinvertebrate communities. In the lower River Murray, approximately 70% of the numerical biomass was represented by four taxa: *Paratya* and *Caridina* (shrimps), *Micronecta* (corixids) and *Chironomus* (chironomid). The atyids (freshwater shrimp) and chironomids (non-biting midges) dominated in all permanently inundated mesohabitats (permanent wetlands, backwater, anabranch, main channel, channel above weir and channel below weir mesohabitats). Only temporary billabongs had a distinct community composition. This demonstrates (i) the dominance of highly connected permanently inundated habitats in the lower River Murray, and (ii) that low temporal variability in flow results in lower temporal diversity in invertebrate communities. However, temporary wetlands within the lower River Murray had communities similar to those occurring in less disturbed MDB river systems (Sheldon and Thoms 2006), suggesting that reinstating variable connection/disconnection phases can contribute to maintaining a more diverse assemblage.

In the lower River Murray the diversity of native gastropods (snails) has decreased from approximately 18 species pre-regulation, to only 1 native species being commonly recorded following river regulation (Sheldon and Walker 1997, Zukowski and Walker 2009). The decline in native species may be due to changes in biofilm composition (Sheldon and Walker 1997), and increases in river/wetland salinity that exceed the tolerance of eggs of some native species (Zukowski and Walker 2009, Evans 1981). Higher fecundity, temperature and salinity tolerance, combined with shorter development time, and competitive behaviour of non-native gastropod species may also contribute to the decline in native species (Zukowski and Walker 2009).

Micro-invertebrate diversity is believed to be derived from the floodplains and sheltered backwaters, not from upstream reservoirs or the main channel of rivers. The progressive loss of intact wetlands due to agriculture, salinisation, or lack of flooding will lead to a loss of the source (i.e. wetland/floodplain seed banks) of microfauna (Brookes et al. 2009). It has been demonstrated (Boulton and Lloyd 1992) that once the duration between floods exceeds 11 years, the diversity of invertebrates present in soil egg banks, and the number of animals hatching once soils are finally inundated decreases significantly. Those authors propose that removing floods that occur at a frequency of <10 years may reduce emergence of invertebrates from the seedbank, reducing floodplain productivity and the abundance of food resources for higher trophic levels including fish. It is unclear what impact grazing, cropping or agriculture on floodplains has on emergence of aquatic invertebrates. However, disruption of the soil profile may reduction is likely to have a negative impact (Boulton and Lloyd 1992).

Lake *et al.* (1989) proposed three phases of community development in newly inundated wetlands.

1. a rapid increase in taxa richness as a result of colonisation,
2. a period of constant taxa richness underscored by a continual turnover of taxa (probably as a result of increasing competition and predation)
3. community dominated by high levels of predation.

This successional pattern is considered to be driven by a range of factors including flood history (Boulton and Lloyd 1992), and the mode of colonisation of different taxa. This includes those that are transported with the source water (transported colonists e.g. rotifers), aerial colonisers (e.g. odonata and trichoptera) (Lake et al. 1989) and those that emerge from a dormant state in the sediments (e.g. chironomids) (Tucker 2006). It is of note that some predators (e.g. trichoptera, economids and ceratopogonids) are highly mobile and fecund with a short life–cycle, thus can rapidly increase in abundance in newly inundated wetlands (Hawking et al. 2005). Some taxa (e.g. trichoptera) have life stages that include flying adults which enable them to rapidly colonise new habitats (Moorhead et al. 1998) providing distances between the source and new habitats are relatively short. Others have dormant stages that remain viable in sediments during extended periods of low or no flow (Danks 2000). These taxa will rapidly emerge in a newly wetted wetland, often being amongst the first to appear after inundation. Jenkins and Boulton (2007) suggest micro–invertebrate emergence can provide a suitable indicator of arid floodplain health. Those authors suggest that micro–invertebrate densities of 100–1000 individuals L^{-1} within 2–3 weeks after flooding indicate intervention success. Assessments of the micro–invertebrate community in Lake Limbra (Chowilla Floodplain) and the adjacent creek system during and following the 2010–11 flood, demonstrate mean densities of 2,104 (±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 (Furst, unpub.).

Impact of increasing river/wetland salinity

Wetland sediments that have been challenged by high salinities may have depauperate freshwater seed banks as adult invertebrates and mature plants may not have survived in the surface waters to deposit seeds or eggs in the sediments (Brock et al. 2005). Species diversity is inversely related to salinity (Brookes et al. 2009). Nielsen et al (2003) report reduced number and abundance of taxa in invertebrate communities when the water used to inundate floodplain soils increased from low (1000 mgL^{-1} ~1,560 μScm^{-1}) to high (5,000 mgL^{-1} ~ 7,810 μScm^{-1}) salinity. Few aquatic invertebrates germinated when the surface water salinity was 5,000 mgL^{-1}. For some wetlands, the threshold for change occurs when salinity increases to between

1,000 and 2,000 mgL^{-1} (Brock et al. 2005). High surface water salinity may influence emergence from the seed and egg bank in three ways; (i) reducing the viability of the dormant eggs and seeds, (ii) interference with germination/emergence cues, or (iii) toxicity effects kill any emergents / germinants (Nielsen et al. 2003).

Role of different water sources in expected outcomes from use of environmental water

The microfauna of water from the Darling and Murray Rivers are markedly different. The community of the River Murray is dominated by lacustrine (limnoplankton) species, including microcrustacea (copepods, cladocerans and ostracods) that require stable conditions and weeks to close the life cycle. In contrast, the Darling River has a riverine (potamoplankton) microfaunal community that is dominated by rotifers (Shiel et al. 1982). Rotifers are able to reproduce within days, allowing them to complete their life cycle very rapidly. The microfaunal communities demonstrate the habitat differences between the heavily regulated River Murray, with its network of lotic in–channel and off–channel habitats, and the moderately regulated, comparatively lentic Darling River. Travel times for regulated flows from the respective storages to Lake Victoria are 14 days from Menindee Lakes (Darling River), 26 days form Hume Dam (Murray River) and 35 days from Burrinjuck Dam (Murrumbidgee River) (data from MDBA). It is of note that the composition of microfauna also varies between storages with short (e.g. Lake Mulwala) and long (e.g. Hume Dam) retention times. The residence time of water within Hume Dam during the period November 2005–April 2007, a period of sustained drought, was estimated at 150-350 days (Cartwright 2010).

3.7. Carbon and nutrients

Carbon in river systems may be comprised of dissolved inorganic carbon (DIC), particulate inorganic carbon (PIC), dissolved organic carbon (DOC) or particulate organic carbon (POC). Major potential sources of DIC in the Murray River include groundwater, rainfall, silicate/carbonate weathering, respiration from soils, and mineralisation of organic material (Cartwright 2010). Natural organic matter (NOM) is a significant energy source in aquatic systems (Wetzel 2001) and originates either from terrestrial sources within the catchments of creeks, rivers and lakes (allochthonous organic matter), or from organisms within the aquatic ecosystem (autochthonous organic matter).

Phytoplankton, periphyton, bacteria, and macrophytes represent the major sources of autochthonous NOM. The primary sources of allochthonous NOM in forested catchments are large woody debris (LWD; logs, branches), coarse particulate material (CPOM; leaf litter, bark, twigs), fine particulate material (FPOM; generated from processing of LWD, CPOM and the flocculation of dissolved organic matter), and dissolved organic matter (DOM) (Robertson *et al.* 1999, Ward 1986). It is generally considered that a substantial component of the allochthonous DOM entering streams via multiple pathways, including overland, sub–surface and downstream flow is leached from terrestrial plant material (McKnight *et al.* 2003, Baldwin 1999). However, soils (Nelson *et al.* 1996), and grazed grasslands (McTiernan *et al.* 2001) are also a key source of DOM in some catchments.

The release of nutrients and carbon from inundated floodplains is well documented (Baldwin and Mitchell 2000) and is believed to be an important component in the functioning of lowland river systems fuelling in-channel productivity (e.g. Robertson *et al.* 1999, Junk *et al.* 1989, Francis and Sheldon 2002). When surface water first enters a floodplain there is a very rapid (i.e. within hours) leaching of carbon and nutrients from floodplain tree leaf litter (Baldwin 1999, Francis and Sheldon 2002, Wallace *et al.* 2008a, O'Connell *et al.* 2000), coupled with a pulse of carbon and nutrients from newly inundated soil – the 'Birch effect' (Scholz *et al.* 2002, Kobayashi *et al.* 2008, Wilson *et al.* 2010, Banach *et al.* 2009).

Studies in lowland rivers in south-eastern Australia

In south-eastern Australia, floodplain eucalypts, particularly river red gum (*Eucalyptus camaldulensis*) and to a lesser extent black box (*E. largiflorens*) generate a large standing biomass of leaf litter (ca. 2,500 gm^{-2} and 600 gm^{-2} respectively (Wallace and Lenon 2010, Wallace 2009). This material represents a large source of allochthonous organic matter to floodplains and wetlands (Francis and Sheldon 2002, O'Connell *et al.* 2000, Wallace *et al.* 2008b, Brookes *et al.* 2007, Hladyz *et al.* 2011, Glazebrook and Robertson 1999a). Leaves of *E. camaldulensis* have a half life of 6–12 months (Glazebrook and Robertson 1999b), with seasonal litter fall (Briggs and Maher 1983) contributing to the long–term maintenance of a large pool of material. With an estimated decay period of 375 years (Mackensen *et al.* 2003), large woody debris such as branches from river red gum is not likely to be a substantial source of carbon to foodwebs. In non–wooded areas wetland/floodplain areas, the standing load of leaf litter has been shown to be ca. 440 gm^{-2} from annual and perennial understory vegetation (Wallace and Lenon 2010, Wallace 2009) and ≤ 750 gm^{-2} from aquatic macrophytes (Baldwin *et al.* 2013). In a study undertaken on the Chowilla Floodplain (South Australia), Hackbusch (2011) demonstrated that the magnitude of DOC released from floodplain soil was significantly lower (approximately 44 times lower) than that released from plant material. Data from other studies undertaken at the same floodplain (Wallace *et al.* 2008a, Wallace and Lenon 2010, Wallace 2009) can be used to demonstrate that ca. 250 tonnes of DOC would be released into the water column per km^2 of river red gum woodland.

Quality of DOC and nutrients for food webs

Biodegradable DOC will be used in days–months (Servais *et al.* 1989). Labile DOC will be used within 1-2 weeks (Sondergaard and Middleboe 1995) and may be oxidised very rapidly (e.g. hours-days) (Wallace *et al.* 2008a), hence is unlikely to persist in ambient DOC pools (Findlay and Sinsabaugh 1999, Imai *et al.* 2001). Within hours to a few days, nitrogen and phosphorus undergo transformation and assimilation by organisms on the floodplain. Phosphorus is assimilated by organisms with the overall movement and uptake of phosphorus dependent on the length of time water remains on the floodplain (Schramm *et al.* 2009). Nitrate is either taken up by microorganisms and algae, or is respired through denitrification such that floodplains act as a sink for N (Forshay and Stanley 2005).

It is generally considered that terrestrial plants and macrophytes that have relatively high carbon:nitrogen (C:N) and carbon:phosphorus (C:P) ratios may not be a significant food resource for higher trophic levels (Bunn *et al.* 2003, Bunn and Boon 1993) unless the energy is processed via the microbial loop (del Giorgio and Cole 1998). The assimilated carbon and nutrients are subsequently cycled though the food web to higher trophic level organisms (e.g. birds and fish) via multiple pathways, including micro– and macro–invertebrates. Sherr and Sherr (1988) propose that the microbial food web is capable of transporting a significant proportion of carbon to zooplankton. This process is referred to as 'trophic upsurge' (Kern and Darwich 1997, Furch and Junk 1997, Geraldes and Boavida 1999, Scharf 2002, Lourantou *et al.* 2007, Talbot *et al.* 2006).

Autochthonous and allochthonous organic carbon

Oliver and Merrick (2006), Oliver and Lorenz (2007) and Gawne *et al.* (2007) demonstrated that the River Murray is energy constrained with net production close to zero. Studies in the Logan, Gwydir and Ovens Rivers (Hadwen *et al.* 2009) and Lachlan River (Moran 2011) have also demonstrated that respiration of the heterotrophic bacterial community and dissolved organic carbon (DOC) consumption is limited by the quality of DOC present. This is considered to be the case for the majority of Australian rivers during low flow conditions when allochthonous DOC supply is limited (Robertson *et al.* 1999).

Robertson *et al.* (1999) suggest primary productivity derived from autochthonous sources (i.e. phytoplankton) is trivial compared with DOC released from leaf litter, and predicted that a flood inundating 44 km^2 would provide as much allochthonous derived carbon as produced from autochthonous sources in one year. Gawne *et al.* (2007) estimate that this effect would be produced from only 34 km^2. Those authors calculated that phytoplankton dominate both primary productivity (57–68%) and respiration (52–66%) and that the contribution of biofilms contributed <10% of all gross primary production and respiration. Conversely, Cartright (2010) conclude that photosynthesis is probably a relatively minor process in the carbon balance. Whitworth and Baldwin (2013) suggest that the relative contribution of biofilms in energy pathways has been reduced due to the comprehensive and systematic de-snagging programs that operated from the mid- 1800s until the early 1990s in rivers of the Murray-Darling Basin. De-snagging removed much of the substrate that biofilms colonise. DOC and nutrients assimilated into biofilms may short-circuit the path from microbial loop → micro-invertebrates → progressive higher trophic levels, as biofilms may be grazed directly by large macro-invertebrates (shrimp) or molluscs (snails) that may in turn be consumed directly by fish (refer Fish above).

Conceptual models

There is often conjecture as to whether allochthonous (externally derived, e.g. leaf litter from riparian trees) or autochthonous (internally derived, e.g. phytoplankton, benthic algae, macrophytes) sources of NOM are most important. The relative role of each source has received a significant amount of attention, summarised in a series of conceptual models emphasizing differing aspects. The river continuum concept (RCC; Vannote *et al.* 1980) focuses on the importance of longitudinal inputs and the downstream dispersion of resources. In this model, there is a switch from allochthonous to autochthonous inputs being the dominant source down the longitudinal gradient from headwater to lower reaches. The serial discontinuity concept (Ward and Stanford 1983) incorporates the influence of barriers (large dams) on the RCC. The flood pulse concept (FPC; Junk *et al.* 1989) extends the RCC and incorporates the importance of lateral connectivity and the two–way transfer of material between the river and its floodplain. During the rising limb of the hydrograph, material is transported from the river to the floodplain. In the recession phase, salt, carbon, nutrients, plankton, propagules and fish are transported from the floodplain to the river. The riverine productivity model (RPM; Thorp and Delong 1994) emphasizes the role of local sources of carbon including both autochthonous production and direct litterfall from the riparian zone. In arid and semi–arid regions, episodic pulses of rain may be a primary driver of plant growth (the pulse reserve model Noy-Meir 1973, Reynolds *et al.* 2004). While each model has application, variability between aquatic systems means no specific model can be expected to be universally applicable (Robertson *et al.* 1999); the types of inputs/outputs and their spatio–temporal variability must be considered (Anderson *et al.* 1996).

It is important to note that none of the conceptual models described above (Junk *et al.* 1989, Vannote *et al.* 1980, Ward and Stanford 1983, Thorp and Delong 1994) are based on Australian river systems. A conceptual model that is designed for lowland rivers in the Southern MDB is currently a key knowledge gap. It is likely that such a model would predict that the dominant source of DOC would shift dynamically as hydrology shifted between periods of low flow, moderate flow and flood periods and back again. During periods where flows are constrained to the river channel, the majority of energy is likely to be derived from in-stream sources, as the exchange of material between the floodplain and the river channel is limited. The relative importance of allochthonous carbon is likely to increase when catchment discharge, lateral connectivity and therefore input of allochthonous material is increased (Tockner *et al.* 2000), providing a short–lived but significant productivity boom. There is a large store of leaf litter deposited annually from floodplain eucalypts, which accumulates between inundation events. Aging (e.g. rainfall, weathering) reduces the amount and bioavailability of DOC released from leaf litter (Baldwin 1999). However, there will be a larger store of nutrients available to be released to the water column as the inter-event duration between flooding increases.

Implications of carbon and nutrients from floodplains for water quality

Primary controls for the concentration of dissolved oxygen (DO) in water are (i) the physico–chemical processes of gas–exchange (oxygen diffusion into the water column across the air–water interface and water–gas saturation relationships); (ii) oxygen production from photosynthesis; (iii) sediment and pelagic oxygen demand (e.g. heterotrophic and autotrophic respiration, nitrification); and (iv) chemical oxygen demand (Kelly 1997, Matlock *et al.* 2003, Burns *et al.* 1996, Odum 1956). When oxygen demanding processes exceed oxygen re–supply via diffusion or photosynthesis, DO concentrations are depleted and hypoxic (<2 mgO_2L^{-1}) (Sabo *et al.* 1999, Roman *et al.* 1993, Davidson *et al.* 1998) and or anoxic (zero oxygen) conditions occur.

Tolerance to hypoxia is species and life-stage specific. Consequently, changes in DO concentration can have significant impacts on biodiversity (Ekau *et al.* 2010). Hypoxia substantially reduces the amount of habitat available to support fish and other aerobic organisms (Turner and Erskine 2005). It is well recognised that hypoxia is associated with fish kills (Erskine *et al.* 2005), and it is generally accepted that fish will begin to suffer at dissolved oxygen concentrations less than 5 mgO_2L^{-1}, and that few species can tolerate conditions of <3 mgO_2L^{-1} for prolonged periods (Ekau *et al.* 2010, Gilligan *et al.* 2009). In warm waters, fish generally die when exposed to DO concentrations lower than 1.5 mgL^{-1} (Miranda *et al.* 2000). Hypoxia is also known to disrupt endocrine systems (Wu *et al.* 2003) embryonic development (Shang and Wu 2004) and survival and hatch rates (Hassell *et al.* 2008) of fish. Larvae and young–of–year juveniles may survive at dissolved oxygen concentrations as low as 20% saturation (e.g. 1.8 mgL^{-1} at 20 ºC), but growth is likely to be restricted (Gehrke 1988, 1991).

Hypoxia has been shown to lead to deformities and death of embryos of some frog species, and hatching at earlier stages of development. Early hatching is likely to have a negative impact on growth, ability to avoid predation and reproductive success of adult stages (Seymour *et al.* 2000, Mills and Barnhart 1999). Degradation of aquatic macroinvertebrate communities is known to occur under hypoxia in streams (Walsh 2002, Walsh *et al.* 2001, Feminella *et al.* 2003) and wetlands (Spieles and Mitsch 2003). The influence of hypoxia on the structure of the micro invertebrate communities in wetlands has not been widely studied but published research (e.g. Roman *et al.* 1993, Davidson *et al.* 1998, Mikschi 1989, Watkins *et al.* in review) suggests community structure would be substantially altered (Davidson *et al.* 1998). An ongoing study has demonstrated that the density of copepod nauplii is depressed over a gradient of hypoxia in floodplain wetlands at Chowilla Floodplain (Furst, unpub.).

Anoxia may also substantially disturb nutrient cycling regimes, with a cascade effect leading to the loss of refugia for river fauna (Heathwaite 2010). Anoxia in the hypolimnion (lower section of the water column) may lead to the release of 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); conversion of dissolved organic nitrogen to ammonia and nitrate (Harris, 2001) and accumulation of redox sensitive compounds from anoxic sediments (e.g. Baldwin and Mitchell, 2000; Dahm *et al.*, 2003) some of which (e.g. ammonium and sulfide) are toxic to many aquatic organisms (Vismann, 1996; Hickey and Martin, 1999). These toxic compounds may then act synergistically with other stressors such as high temperature and/or low dissolved oxygen to cause fish deaths (Lloyd 1961, Sargent and Galat 2002).

Blackwater events

Conditions where the surface water contains enough dissolved organic carbon (DOC) to discolour the water sufficiently to resemble dark "tea" are referred to as blackwater events. These events are often, but not always, associated with low dissolved oxygen (DO) concentrations (Howitt *et al.* 2007, Meyer and Edwards 1990). Blackwater events, and the biogeochemical cycles involved are natural processes (Baldwin and Wallace 2009).

Data collected from managed wetlands on the Chowilla Floodplain (Wallace and Lenon 2010, Wallace 2008) demonstrates that it is possible for the concentration of oxygen in the water within impounded areas to decline from normoxic (~8 mg O_2 L^{-1}) to the trigger level for stress for native fish (4 mgO_2L^{-1}) within 2 ½ days, and to the level where fish deaths may be expected (~2 mg O_2 L^{-1}) within an additional 1 ½ days. Those authors describe four key phases in dissolved oxygen dynamics: (1) *Filling phase*; wetland is supplied with water and leaching of carbon and nutrients from inundated plant material and floodplain soil occurs rapidly; (2) *Initial oxygen depletion period*; heterotrophic metabolism of carbon and nutrients is the dominant process. Depending on the morphology and size of the wetland, this phase may be initiated very rapidly upon filling, or at a delayed time, once water levels have reached the fringing tree line and inundated the accumulated floodplain leaf litter; (3) *Alternating states*; there are rapid diurnal changes in oxygen concentration driven by alternate processes – production of oxygen by photoautotrophs (and subsequently increasing oxygen concentrations during the day) and consumption of oxygen by heterotrophs and autotrophic respiration (leading to hypoxic/anoxic conditions at night); (4) *Equilibrium*; the majority of the labile organic material has been consumed. Consumption of recalcitrant material and internal cycling of material within the microbial loop is likely to be occurring, but the system is likely to become either slightly heterotrophic or autotrophic due to the increased nutrient concentrations.

The managed flooding of Barmah Forest in 2000 released a pulse of hypoxic (dissolved oxygen ≤2 mg O_2 L^{-1}) water back to the Murray and Edward Rivers that led to significant fish-mortality (Howitt *et al.* 2007). The 2010-11 flood in the Murray system was also characterised by a system wide blackwater event distinguished by high DOC and low dissolved oxygen. That event resulted in hypoxic conditions stretching from the Barmah–Millewa Forest to the Lower Lakes, including the lowland reaches of all major tributaries downstream of Yarrawonga Weir. Widespread fish kills resulted (Whitworth *et al.* 2011).

Flows released into creek and river systems in order to reinstate downstream supply have been associated with hypoxic blackwater events. Major fish deaths were reported in association with a replenishment flow in Colligen and Merrin Creeks (part of the Wakool anabranch system, NSW) in January–February 2009 (Baldwin and Whitworth 2009), and in the Darling River during February 2004 (Ellis and Meredith 2004). Hypoxic conditions were recorded in association with replenishment flows in the Darling River in 2007–08 (Wallace *et al.* 2008b) and the Lachlan River in 2010–11 (Wallace and Bindokas 2011). The relatively slow (i.e. days) decline in DO recorded in association with the arrival of flow fronts in the Lachlan River were considered to be driven by increases in *in–situ* heterotrophic activity stimulated by an increase in bioavailable dissolved organic carbon (DOC) resulting from the catchment derived inflows (Wallace and Bindokas 2011). In contrast, the immediate depletion of DO recorded in association with the arrival of flows in the Darling River suggest that oxygen depletion was driven by either chemical oxygen demand resulting from the rapid oxidation of reduced sulphur compounds associated with re–suspension of sulfidic sediments (e.g. Baldwin and Whitworth 2009, Sullivan *et al.* 2002, Johnston *et al.* 2003); or from the downstream passage of a discrete body of hypoxic/anoxic water at the head of the flow (Wallace *et al.* 2008b). Similarly, flows along dry river channels in the Wakool anabranch system mobilised a substantial load of organic debris (i.e. leaf litter, bark) creating very low dissolved oxygen concentrations (e.g. 0.6 mgL^{-1}) in refuge pools that had previously been habitable (Gilligan *et al.* 2009).

Management of blackwater events

In rivers and creeks used as part of the water supply network, leaf litter will accumulate in the channel during enforced low- or no-flow periods. On floodplains where the frequency and duration of inundation is reduced due to catchment development and flow regulation, several years worth of leaf litter and terrestrial vegetation may build up. Delivery of water to reinstate flow (i.e. to deliver water to downstream users or improve water quality in residual pools), or create a managed inundation of a floodplain that occurs in summer coincides with (i) the peak litter fall period and (ii) high water temperature. Furthermore, during managed flooding or managed flow releases, the volume and exchange of water is likely to be substantially lower than would occur under pre-regulation conditions. All of these factors increase the likelihood of onset of hypoxic conditions. Although fish deaths associated with hypoxic blackwater events would have occurred pre-regulation, the frequency of risk is considered to have increased as a result of regulation. In addition, the natural level of resilience and ability of ecological communities to withstand and recover from such events has been reduced due to barriers to movement, overfishing, etc.

A number of factors are critical in determining whether or not a blackwater event will result in a fish kill. The two most important factors are water temperature and carbon loading (Baldwin and Wallace 2009). Organic loading (amount of carbon and nutrients and the stoichiometry of those nutrients) in water overlying floodplains is dependent on vegetation type and condition (Wallace *et al.* 2008a, Wallace 2009, Brookes *et al.* 2007) flood timing (Baldwin 1999, Watkins *et al.* 2010a, Watkins *et al.* 2010b) and whether or not the accumulated litter has been flooded before (O'Connell *et al.* 2000). Flooding in late spring and summer is considered problematic (Baldwin and Wallace 2009) as (i) peak litter fall for eucalypts occurs in summer (Briggs and Maher 1983) and (ii) for every 10 °C increase in water temperature the rate of oxygen depletion approximately doubles (Howitt *et al.* 2007).

The risk of establishment of a blackwater event can be largely managed by (i) not using ponded floods for delivery of environmental water allocations; (ii) maximising water exchange when using large constructed infrastructure; (iii) avoiding flooding during warm periods; (iv) use dilution flows from water sources with normoxic oxygen and relatively low concentrations of DOC to dilute hypoxic blackwater; and (v) increase the frequency of flooding and control timing of flooding such that inundation occurs prior to peaks in litter accumulation (Baldwin and Wallace 2009, Whitworth *et al.* 2011).

Implications for potable water supply

In catchments where water is harvested for potable supply, alterations to the composition and bioavailability of allochthonous DOC entering aquatic ecosystems may be highly problematic. DOC can function as the principal factor influencing treatment cost (Hine and Bursill 1987). NOM fouls membrane filters used in potable water treatment (Cho *et al.* 1999), and some DOC fractions (e.g. those with neutral charge) are resistant to traditional pre–treatment techniques (Chow *et al.* 2000, Chow *et al.* 2004). These fractions react with chlorine during the disinfection phase, to generate potentially toxic or carcinogenic by–products, and contribute to bacterial regrowth in distribution systems (Prevost *et al.* 1998, Simpson and Hayes 1998). Furthermore, DOC complexes with metals (e.g. lead and zinc) and hydrophobic compounds such as hydrocarbons, herbicides and pesticides, and the transport of DOC and toxic pollutants from surface runoff into ground water via infiltration systems is of substantial environmental concern (Ellis and Hvitved-Jacobsen 1996). Consequently, the characterisation of DOC to improve our understanding on the composition and reactivity of DOC is of strong interest to the water science and technology community (Chow *et al.* 2004, Weishaar *et al.* 2003).

3.8. Algal blooms

In the lower River Murray, algal blooms have been recognised as problematic for 135 years. Scums of algae were reported to have occurred in the lower River Murray in 1853, 1878, 1880 and 1888. In 1878, several hundred stock were poisoned at Lake Alexandrina after drinking from the lake during a bloom of the cyanobacteria *Nodularia spumigera* that had developed during a period of low river, low lake water levels and high temperatures. It is this poisoning event which raised first raised public awareness of the problems associated with cyanobacterial blooms. Problematic algal blooms occurred again in the lake in the early 1990's (Codd *et al.*, 1994). Following an extensive cyanobacterial bloom (*Anabaena circinalis*) that stretched for more than 1,000 km in the Darling-Barwon River in 1991 (Bowling and Baker, 1996), the Murray Darling Basin Ministerial Council commissioned the development of an algal management strategy (MDBC, 1993). That event was the largest riverine cyanobacterial bloom ever recorded (Maier *et al.*, 2001). Blooms in the River Murray in South Australia are typically dominated by *A. circinalis* and occur in summer during low flow conditions (Burch *et al.*, 1994). Such blooms are problematic due to impacts on users of untreated water (i.e. irrigators, graziers), recreational users of the river and lakes, and the pipelines that convey water from the river to the city of Adelaide and other regional centres.

The hazards associated with cyanobacteria range from public health related toxicity issues to aesthetic water taste and odour issues. The toxins produced by cyanobacteria include both 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 compounds are difficult to remove with conventional water treatment and require expensive activated carbon for adequate removal. Consequently it is important to minimise cyanobacterial biomass in the river to reduce the risk from toxins and the taste and odour compounds.

A range of parameters are involved in determining the likelihood, onset and duration of cyanobacterial blooms. Factors include water temperature, light availability, nutrient availability, 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). The onset of thermal stratification is a function of river flow, solar radiation and wind speed (Bormans *et al.*, 1997; Maier *et al.*, 2001). Although persistent thermal stratification in rivers is not a typical occurrence because discharge normally provides sufficient kinetic energy to prevent its establishment, it can occur within deep (e.g. >5m) pools during low and no–flow periods in summer (e.g. Turner and Erskine, 2005 and references therein). Sherman et al., (1998) demonstrated that in the Murrumbidgee River (NSW, Australia) the median euphotic depth during the 1993–94 summer was 1.45 m, and that the establishment of persistent stratification at approximately 2 m caused a transition between the two dominant phytoplankton species. This was due to the lack of water column mixing to entrain non–motile (or species that cannot control their buoyancy) phytoplankton in the water column. Under the low–flow conditions experienced, negatively buoyant species such as the diatom *Aulacoseira* spp. rapidly sunk out of the euphotic zone, and buoyant cyanobacteria species such as *Anabaena* spp. become dominant.

Lower River Murray studies

In the lower River Murray, the combination of ten low level weirs, gentle gradient (50mm per km^{-1}), low summer flows, and tightly controlled river management has produced a shift from highly variable riverine conditions to stable conditions more representative of a string of interconnected lakes (Walker and Thoms, 1993; Baker *et al.*, 2000). Low flow conditions in summer may favour buoyant cyanobacteria rather than diatoms and other phytoplankton groups. Buoyancy is a potential light harvesting advantage in turbid stratified water bodies (Baker *et al.*, 2000). However, conditions are not always conducive to the establishment of a bloom due to a combination of discharge and wind speed.

Bormans et al., (1997) calculated that persistently stratified conditions can develop when flow velocities are < 0.1 ms^{-1}. During periods of entitlement flow in the lower reaches of the South Australian River Murray when water velocity is low (0.04-0.06 ms^{-1}) and solar radiation is high (i.e. summer), wind speed is the dominant factor limiting the development and persistence of thermal stratification (Bormans *et al.*, 1997; Maier *et al.*, 2001). During periods of entitlement flow, the depth of the mixed layer can vary from one side of the river to the other in some reaches due to protection from wind driven mixing and shading provided by high cliffs that characterise some reaches (Bormans *et al.*, 1997). The approach used by those authors to determine the likelihood of stratification could be used to ascertain the increased likelihood of persistent stratification (i) associated with large magnitude weir pool raisings during periods of low flow, or (ii) resulting from reinstating normal weir pools heights after weir pool lowering.

Stratification that persists for 2 days is sufficient for existing cells of buoyant species such as *A. circinalis* to accumulate in the surface layer, and for short lived blooms to become problematic. The magnitude of the problem increases markedly if the water column remains stratified for more than 7 days, as these conditions could result in substantial growth of the population and a sustained bloom (Maier *et al.*, 2001). Those authors calculated that the probability of thermal stratification existing for 2 days under entitlement flow conditions is highest in January (45%) and December (39%). In contrast, probabilities of thermal stratification existing for 7 days are low; <1% in January and <2% in December. Furthermore, the recurrence interval for these conditions is 50 years. In terms of management options, increased flow generated by releases from upstream storages (e.g. Lake Victoria) substantially reduces the probability of sustained stratification. Despite this, the maximum release from Lake Victoria is not sufficient to completely eliminate the potential for persistent stratification. In addition, this is a costly management tool. Based on a cost of $50 per ML, a release of 10,000 MLday^{-1} from Lake Victoria for 7 days would cost ca. $5 Million. It is of note that a reduction in weir pool level of 300 mm at Locks 1-3 has a slightly larger impact on reducing the probability of sustained thermal stratification than the release of an additional 10,000 MLday^{-1}. Those authors concluded that this is the cheapest and most effective option for bloom management.

Studies have demonstrated that the lower River Murray is nitrogen limited, and that conditions may favour heterocystic nitrogen fixing species of cyanobacteria (Baker *et al.*, 2000). Water sourced from the Darling River typically has higher nutrient loads than water from the upper Murray. However, any impact of the higher nutrient load may be at least partially offset by higher turbidity and reduced depth of the euphotic zone (Baker *et al.*, 2000). It must be noted that the release of nutrients from the inundation of accumulated material on floodplains will produce a nutrient pulse capable of supporting significant phytoplankton biomass (Brookes *et al.*, 2007; Wallace, 2008; Wallace and Lenon, 2010). This could be problematic during managed inundations using constructed infrastructure during periods of low flow (Brookes *et al.*, 2007). Baker et al., (2000) demonstrated that phytoplankton move laterally at the same rate as the water in the river. Therefore, slow flowing areas or zones with long retention times may increase the biomass of algae in the river by allowing additional growth. In addition to phytoplankton cells that have been derived from upstream sites and in-channel growth, seed populations of phytoplankton may be entrained into the river channel from the floodplain and/or connected wetlands, particularly following rapid drops in water level. However, depending on prevailing conditions, localised blooms may not persist for long distances downstream as the river is generally not conducive to sustaining blooms (Bormans *et al.*, 1997).

Conceptual models

Maier et al., (1998) developed an artificial neural network for predicting the growth of *A. circinalis* at Morgan. The dominant variables in the model are flow, temperature and colour. All of these physical parameters affect the stratification behaviour of the water column. The concentration of phosphorus, nitrogen and iron was less important than the physical parameters. Bormans et al., (1997) demonstrate the important interaction of flow and wind speed in the lower River Murray.

A conceptual model developed by Baker et al., (2000) for the Lower River Murray proposes that during periods of moderate flows, flow is the dominant factor that affects the development of cyanobacterial blooms. Wind speed and source of water are the next most important environmental factors. The model predicts that when flows are:

- >10,000 MLday^{-1}, high turbulence will favour species such as the negatively buoyant diatom *Aulacoseira granulata*.
- ca 10,000 MLday^{-1}, diurnal stratification may occur if wind speed is low (<1.2 ms^{-1})
- ca 4,000 MLday^{-1},
    - persistent stratification may develop at wind speeds <1.2 ms^{-1}
    - diurnal stratification may occur at wind speeds 1.3-3.0 ms^{-1}
- wind speeds >3ms^{-1} will disrupt stratification irrespective of flow
- if water is sourced from the Darling River or upstream flooding, phosphorus concentrations may be elevated, and nitrogen concentrations will determine the dominant phytoplankton species.

Trophic cascades

Multiple pathways for the cycling of nutrients (Brookes *et al.*, 2005) may provide a mechanism to prevent, or minimise, the occurrence/magnitude of harmful/nuisance algal blooms. Macrophytes are an obvious alternative pathway for nutrient interception. However, in wetlands that have dried for >12 months between floods, it is likely that there will have been a shift from flood dependent macrophytes towards drought tolerant plants (Nicol *et al.* 2010a) that will senesce and release additional nutrients upon inundation. It has been proposed (Polis *et al.*, 2000) that community level trophic cascades can only be effective if each level contains a high proportion of resources that are available to the next level. A high abundance of visually feeding zooplanktivores consuming large, herbivorous zooplankton coupled with stable warm conditions, may lead to nutrients being 'funnelled' into a high relative proportion of inedible cyanobacteria. As a result, the transfer of energy to higher levels via small body zooplanktivorous or herbivorous fish may be inhibited.

In addition to issues associated with algal blooms, thermal stratification is likely to have direct impacts on dissolved oxygen (DO), plankton and solutes (Becker *et al.*, 2010). The onset of persistent stratification isolates the hypolimnion (bottom layer of the water column) and precludes supply of oxygen from the air–water interface reaching the bottom layer. Deoxygenation of the hypolimnion will substantially reduce the amount of habitat available to support fish and other aerobic organisms (Turner and Erskine, 2005) and lead to poor water quality conditions due to the release of sediment bound material (see discussion in previous sub–section). During periods of low water availability, river operators and environmental managers may choose to use "sustaining" flows (Lind *et al.*, 2007) to disrupt thermal stratification and break up cyanobacterial blooms (Maier *et al.*, 2001). However, such flows may create hypoxic flow fronts that are problematic to fish and other aquatic organisms (refer Aquatic macro- and micro-invertebrates are a large, heterogeneous group of organisms that includes crustaceans, insects, worms, rotifers, leaches, molluscs, decapods and sponges (Brookes et al. 2009). Incorporating herbivores, detritivores and predators, macro– and micro–invertebrates are important consumers within wetland ecosystems, incorporating a wide range of functional groups (*e.g.* filtering collectors, predators, scrapers). They also provide a vital pathway in trophic cascades between bacterial and algal production and the higher order trophic levels (Bunn et al. 2003). By consuming bacteria and microalgae, they indirectly alter nutrient cycles, transfer energy to secondary consumers including macro-invertebrates, fish and waterbirds (Jenkins and Boulton 2003). Many waterbirds feed extensively on micro-invertebrates (Crome 1985). Invertebrates are a key food resource for breeding waterfowl as they provide the protein source required for egg and nestling development. The responses of waterbird guilds that are piscivorous, herbivorous, reliant on aquatic macro–invertebrates 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). The increase in habitat and food resources (the productivity boom; Bunn *et al.* 2006) associated with floods is considered to provide a direct benefit for native fish (Harris and Gehrke 1994a, Bunn and Arthington 2002, Arthington et al. 2005), many of which may rely on micro-invertebrates for their first feed (see Jenkins and Boulton 2003).

Invertebrate community composition is influenced by many factors including water and habitat quality and quantity, linkages between wetland and riverine environments and trophic interactions (Reid and Brooks 2000). Increasing (i) habitat complexity, (ii) protection or cover from predation and (iii) diversity of food sources for a range of many functional feeding groups, may enable a variety of different functional feeding groups to co–occur (Boulton and Brock 1999). Many invertebrate taxa are known to be sensitive to water quality and habitat disturbance. Because of this sensitivity and their often rapid response to environmental change, invertebrates are frequently used as an indicator of river health and environmental disturbance (Reid and Brooks 2000, Jenkins et al. 2005).

Role of hydrology

At the basin scale, the heterogeneity of habitats is considered to be responsible for the wide diversity of the micro- and macro-invertebrate community of the MDB (Hillman and Quinn 2002). Hydrology is also a major determinant of invertebrate community composition through its influence on habitat availability and structure (*e.g.* macrophytes) (Puckridge et al. 1997, Brauns et al. 2008). In Australia, extreme temporal flow variability (Puckridge et al. 1998) and episodic lateral connectivity with broad floodplains is believed to stimulate episodic productivity 'booms' (McComb and Qiu 1998). Inundation of temporary wetlands creates conditions substantially more productive than those found in permanent wetlands.

Flow regulation promotes the abundance of some habitat types at the expense of others. For example, in the lower River Murray, it is considered that the high abundance of submerged and emergent vegetation that currently exists is an artefact of river regulation, as photographs taken prior to construction of the locks and weirs indicate that vegetation within the channel was sparse (see Sheldon and Walker 1998). The proliferation of vegetation within the channel may provide some substitution for the decline in floodplain wetlands. However, this will have altered the temporal and spatial distribution of invertebrates (Sheldon and Walker 1997).

In some reaches, construction and operation of in-stream structures and modification of the flow regime may decrease habitat fragmentation by creating permanent connection between macrohabitats (e.g. river channel, anabranch, wetland), that would otherwise only intermittently be connected. In other reaches, regulation may increase fragmentation by decreasing the frequency at which floodplain habitats are inundated (Sheldon and Thoms 2006). Sheldon and Walker (1998) demonstrate the importance of macrohabitats, mesohabitat (pool, riffle) and microhabitats (snag, aquatic macrophytes) in shaping macroinvertebrate communities. In the lower River Murray, approximately 70% of the numerical biomass was represented by four taxa: *Paratya* and *Caridina* (shrimps), *Micronecta* (corixids) and *Chironomus* (chironomid). The atyids (freshwater shrimp) and chironomids (non-biting midges) dominated in all permanently inundated mesohabitats (permanent wetlands, backwater, anabranch, main channel, channel above weir and channel below weir mesohabitats). Only temporary billabongs had a distinct community composition. This demonstrates (i) the dominance of highly connected permanently inundated habitats in the lower River Murray, and (ii) that low temporal variability in flow results in lower temporal diversity in invertebrate communities. However, temporary wetlands within the lower River Murray had communities similar to those occurring in less disturbed MDB river systems (Sheldon and Thoms 2006), suggesting that reinstating variable connection/disconnection phases can contribute to maintaining a more diverse assemblage.

In the lower River Murray the diversity of native gastropods (snails) has decreased from approximately 18 species pre-regulation, to only 1 native species being commonly recorded following river regulation (Sheldon and Walker 1997, Zukowski and Walker 2009). The decline in native species may be due to changes in biofilm composition (Sheldon and Walker 1997), and increases in river/wetland salinity that exceed the tolerance of eggs of some native species (Zukowski and Walker 2009, Evans 1981). Higher fecundity, temperature and salinity tolerance, combined with shorter development time, and competitive behaviour of non-native gastropod species may also contribute to the decline in native species (Zukowski and Walker 2009).

Micro-invertebrate diversity is believed to be derived from the floodplains and sheltered backwaters, not from upstream reservoirs or the main channel of rivers. The progressive loss of intact wetlands due to agriculture, salinisation, or lack of flooding will lead to a loss of the source (i.e. wetland/floodplain seed banks) of microfauna (Brookes et al. 2009). It has been demonstrated (Boulton and Lloyd 1992) that once the duration between floods exceeds 11 years, the diversity of invertebrates present in soil egg banks, and the number of animals hatching once soils are finally inundated decreases significantly. Those authors propose that removing floods that occur at a frequency of <10 years may reduce emergence of invertebrates from the seedbank, reducing floodplain productivity and the abundance of food resources for higher trophic levels including fish. It is unclear what impact grazing, cropping or agriculture on floodplains has on emergence of aquatic invertebrates. However, disruption of the soil profile may reduction is likely to have a negative impact (Boulton and Lloyd 1992).

Lake *et al.* (1989) proposed three phases of community development in newly inundated wetlands.

4. a rapid increase in taxa richness as a result of colonisation,
5. a period of constant taxa richness underscored by a continual turnover of taxa (probably as a result of increasing competition and predation)

6. community dominated by high levels of predation.

This successional pattern is considered to be driven by a range of factors including flood history (Boulton and Lloyd 1992), and the mode of colonisation of different taxa. This includes those that are transported with the source water (transported colonists e.g. rotifers), aerial colonisers (e.g. odonata and trichoptera) (Lake et al. 1989) and those that emerge from a dormant state in the sediments (e.g. chironomids) (Tucker 2006). It is of note that some predators (e.g. trichoptera, economids and ceratopogonids) are highly mobile and fecund with a short life–cycle, thus can rapidly increase in abundance in newly inundated wetlands (Hawking et al. 2005). Some taxa (e.g. trichoptera) have life stages that include flying adults which enable them to rapidly colonise new habitats (Moorhead et al. 1998) providing distances between the source and new habitats are relatively short. Others have dormant stages that remain viable in sediments during extended periods of low or no flow (Danks 2000). These taxa will rapidly emerge in a newly wetted wetland, often being amongst the first to appear after inundation. Jenkins and Boulton (2007) suggest micro–invertebrate emergence can provide a suitable indicator of arid floodplain health. Those authors suggest that micro–invertebrate densities of 100–1000 individuals L^{-1} within 2–3 weeks after flooding indicate intervention success. Assessments of the micro–invertebrate community in Lake Limbra (Chowilla Floodplain) and the adjacent creek system during and following the 2010–11 flood, demonstrate mean densities of 2,104 (±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 (Furst, unpub.).

Impact of increasing river/wetland salinity

Wetland sediments that have been challenged by high salinities may have depauperate freshwater seed banks as adult invertebrates and mature plants may not have survived in the surface waters to deposit seeds or eggs in the sediments (Brock et al. 2005). Species diversity is inversely related to salinity (Brookes et al. 2009). Nielsen et al (2003) report reduced number and abundance of taxa in invertebrate communities when the water used to inundate floodplain soils increased from low (1000 mgL^{-1} ~1,560 μScm^{-1}) to high (5,000 mgL^{-1} ~ 7,810 μScm^{-1}) salinity. Few aquatic invertebrates germinated when the surface water salinity was 5,000 mgL^{-1}. For some wetlands, the threshold for change occurs when salinity increases to between 1,000 and 2,000 mgL^{-1} (Brock et al. 2005). High surface water salinity may influence emergence from the seed and egg bank in three ways; (i) reducing the viability of the dormant eggs and seeds, (ii) interference with germination/emergence cues, or (iii) toxicity effects kill any emergents / germinants (Nielsen et al. 2003).

Role of different water sources in expected outcomes from use of environmental water

The microfauna of water from the Darling and Murray Rivers are markedly different. The community of the River Murray is dominated by lacustrine (limnoplankton) species, including microcrustacea (copepods, cladocerans and ostracods) that require stable conditions and weeks to close the life cycle. In contrast, the Darling River has a riverine (potamoplankton) microfaunal community that is dominated by rotifers (Shiel et al. 1982). Rotifers are able to reproduce within days, allowing them to complete their life cycle very rapidly. The microfaunal communities demonstrate the habitat differences between the heavily regulated River Murray, with its network of lotic in–channel and off–channel habitats, and the moderately regulated, comparatively lentic Darling River. Travel times for regulated flows from the respective storages to Lake Victoria are 14 days from Menindee Lakes (Darling River), 26 days form Hume Dam (Murray River) and 35 days from Burrinjuck Dam (Murrumbidgee River) (data from MDBA). It is of note that the composition of microfauna also varies between storages with short (e.g. Lake Mulwala) and long (e.g. Hume Dam) retention times. The residence time of water within Hume Dam during the period November 2005–April 2007, a period of sustained drought, was estimated at 150-350 days (Cartwright 2010).

Carbon and nutrients above).

3.9. Biofilms

Biofilms are a layer of microorganisms that adhere to surfaces including inorganic (e.g. rocks, gravel, sediment) and organic (e.g. wood, aquatic macrophytes) substrates. Biofilms serve an important biogeochemical role in aquatic systems by altering nutrient cycles and providing food resources to invertebrates and higher order animals (Watts *et al.* 2009a, Vink *et al.* 2005, Burns and Walker 2000a). They have short generation times and are highly responsive to changes in the flow regime over short-temporal cycles (Watts *et al.* 2009a). Light and flow regime are regarded as key drivers controlling succession patterns in biofilms (Watts *et al.* 2009a). In low light and low flow conditions, biofilms are typically dominated by heterotrophs (bacteria and fungi). Where biofilms are within the euphotic zone, autotrophs (algae) typically dominate the biofilm (Lock *et al.* 1984, Burns and Ryder 2001).

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

DOC and nutrients assimilated into biofilms may short-circuit the path from microbial loop → micro-invertebrates → progressive higher trophic levels, as biofilms may be grazed directly by large macro-invertebrates (shrimp) or molluscs (snails) that may in turn be consumed directly by fish. It is considered that late successional biofilms will be less diverse, and have a poorer nutritional value (e.g. a higher C:N ratio) as a food resource than early or mid-stage biofilms. C:N ratios increase between diatoms (bacillariophyceae), non-filamentous and filamentous green (chlorophytes) algae. High C:N ratios (≥10:1) in biofilms may reflect dominance of filamentous green algae (Burns and Walker 2000a). However, results need to be interpreted with due caution. Cyanophyta have low C:N ratios than chlorophytes due to the absence of structural tissue in cyanophytes (Burns and Walker 2000a, Sheldon and Walker 1997). It is considered that algal dominated biofilms are unable to provide a nutritionally adequate food resource to support growth and reproduction for some biota including gastropod snails (Sheldon and Walker 1997, Stevens 2005, Wishart 1994).

Influence of river regulation on biofilms

It is believed that a shift from heterotrophic to autotrophic dominance of biofilms follows the reduction in variation water levels (magnitude and frequency) associated with river regulation, and that there has been a major impact in the lower River Murray. This is because (i) in turbid systems, small changes in water level have a large impact on the euphotic depth and hence 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 is typically much lower than would have been occurred for the same discharge under pre-regulation conditions. Whitworth and Baldwin (2013) suggest that the relative contribution of biofilms in energy pathways has been reduced due to the comprehensive and systematic de-snagging programs that operated from the mid- 1800s until the early 1990s in rivers of the Murray-Darling Basin. De-snagging removed much of the substrate that biofilms colonise.

In a study undertaken in the Murrumbidgee River (NSW), Ryder (2004) demonstrated that biofilms exposed to a high frequency of disturbance (inundated for 5 days and exposed for 9 days successively over a 75 day period) produced 10 x lower biomass, and a heterotrophic biofilm assemblage with an algal community characterised early successional algae compared to biofilms exposed to intermediate (inundated for 11 days and exposed for 21 days successively) and low frequency (75 continuous days of submergence at stable depth) disturbance. Biofilms exposed to the intermediate regime were the most productive. Wallace (2013) provides 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.

Lower River Murray studies

In a study undertaken in Lock 1, Burns and Walker (2000b) demonstrated that the variation in amplitude and duration of water level in the tail water inherent during routine weir operations, was not sufficient to support maintenance of early-mid successional biofilms across most of the water column in either the tail water or weir pool. Those authors concluded that the magnitude and duration of flow variability for ecological outcomes need to exceed the desiccation tolerances of biofilm organisms.

Stevens (2005) demonstrated that biofilms with a low C:N ratio (6:1) allowed offspring of the native river snail *Notopala sublineata hanleyi* to grow faster than biofilms with higher 10:1 and 14:1 C:N ratios. Biofilms sampled from the river from shallow depths (<0.5m) displayed elevated C:N ratios (>10:1). Biofilms from 1m had even higher C:N ratios. Alteration of food quality via river regulation may have caused the near extinction of this species.

Souter *et al.*, (2012) investigated the impact of a weir pool surcharge (raising) that occurred at Lock 5 during 2005-06. The surcharge involved a mean increase in water level of 430 mm spanning the period early October to mid December 2005. It was observed that biofilms that developed during the weir pool raising period remained in what was described as a mid-successional community for a longer period than biofilms that developed during the control period. However, the results did not show an impact due to increased submergence depth or flow velocity. It was concluded that other factors associated with the weir pool raising (e.g. changes in nutrients, light, temperature) must have been the drivers for the differences observed.

Expected influence of environmental management actions

Prior to river regulation, levels in the river were characterised by sustained rises and falls that occurred over periods spanning days-weeks. Under current standard operating regimes, the weirs are managed in response to changes in flow to provide relatively stable water levels (± 50 mm) upstream of the structures. The extent of influence diminishes with distance upstream, such that in the tail waters of the weir pools, levels may fluctuate by ±200 mm on a daily basis (Maheshwari *et al.* 1995). Management actions that increase the variability in conditions may achieve desired ecological outcomes.

Re-operation of existing infrastructure (i.e. weir pool manipulation) to create greater variation in water levels within 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 in the lower River Murray. The main pathways through which environmental flows could be anticipated to alter biofilm composition in the lower River Murray are outlined below:

1. Increases in flow will result in an increase in depth of submergence. However, in order to attain maximum benefit, re-operation of the weirs would be required such that the height of the weir pool is not maintained at normal operating levels. If weirs were maintained at standard operating heights, any benefit is likely to be limited to the tail water zones This would mean responses would be localised, and may not be large enough to have a positive effect at the reach scale

2. Weir pool draw-downs will expose submerged woody material and other substrates that biofilms have colonised. Periods of exposure need to be long and frequent enough for desiccation to alter biofilm composition

3. Changes in flow generated by (i) releases from upstream storages, (ii) periods of unregulated flow generated by rain events, or (iii) return flows from upstream management actions may alter carbon and nutrient concentration and/or bioavailability or other water quality parameters such as turbidity

4. It is recognised that (i) flow velocities of ≥ 0.3 ms^{-1} can influence biomass and taxonomic composition in lowland rivers (Ryder *et al.* 2006), and (ii) that physical abrasion from suspended sediments is important in the scouring of biofilms (Grimm and Fisher 1989). Therefore, periods of increased discharge that result in increased velocity and or sediment load may contribute to scouring of biofilms.

Several studies undertaken in the Mitta Mitta River (Victoria), the results of which are summarised by Watts et al (2009a, 2009b), indicate that the benefits derived from high velocity flows are short lived with communities reverting back to pre-flood communities within days-weeks. In systems that have been held stable for a long period, a large disturbance may be required to allow a return to early successional biofilms (Burns and Ryder 2001). Consequently, managing flows on an intermittent basis for temporary increases in velocity may not be an effective method to improve biofilm condition in the lower River Murray. Raising the height of weir pools during periods of increased flow to maximise the depth of submergence of biofilms may be counterproductive under some circumstances. This may occur if the increase in flow velocity that would normally occur with an increase in flow is muted by the increased storage capacity of the weir pool.

The Darling River is usually highly turbid (>100 NTU), and the euphotic depth is normally 0.6-0.7 m, increasing to 1.5-2.0 m as turbidity decreases (Sherman *et al.* 1998). In contrast turbidity in the lower Murray is typically lower. Median turbidity in the reach of the lock 10 weir pool just upstream of the junction of the Darling is <25 NTU. Downstream of the junction, median turbidity increases to ~60 NTU (Mackay and Eastburn 1990). The shallow euphotic depth (depth at which 1% of incident light is remaining) associated with high turbidity infers that small changes in water level will substantially alter the light environment experienced by biofilms. Therefore, in addition to utilising changes in flow or weir-pool operations to alter the light environment experienced by biofilms, the relative proportion of flow into South Australia derived from (i) the headwater storages in the Murray system, or (ii) flows derived from the Menindee Lakes Scheme, may be used to substantially alter the depth of the euphotic zone

3.10. Drought and salinity

It has been demonstrated (Boulton and Lloyd, 1992) that once the duration between floods exceeds 11 years, the diversity of invertebrates present in soil egg banks, and the number of animals hatching once soils are finally inundated decreases significantly. Wetland sediments that have been challenged by high salinities may have depauperate freshwater seed banks as adult invertebrates and mature plants may not have survived in the surface waters to deposit seeds or eggs in the sediments (Brock *et al.*, 2005). Alternatively, the viable seed and eggs deposited may have exceeded their life–span. Species diversity is inversely related to salinity (Brookes *et al.*, 2009).

Although seed and egg banks are generally long lived, a significant decline (~30 %) in viable seed will occur in drought conditions lasting longer than 5 years (Brock, 2011). Furthermore, it has been demonstrated that increasing surface water salinity has a negative impact on the community composition of aquatic plants and invertebrates emerging from wetland seed banks (Nielsen *et al.*, 2003). Those authors detected a significant change in aquatic plant community when floodplain soils were wetted using low salinity (300 mgL^{-1} or ~470 μScm^{-1}), and high salinity (5,000 mgL^{-1} or ~7,810 μScm^{-1}) surface water. Distinctly different invertebrate communities were recorded with low and moderate (1000 mgL^{-1} or ~1,560 μScm^{-1}) salinity surface water. When surface water salinity increased from 1,000 to 5,000 mgL^{-1}, the number and abundance of taxa of both aquatic plants and invertebrates decreased. Few aquatic invertebrates or plants were found to be capable of germinating when the surface water salinity was 5,000 mgL^{-1}. For some wetlands, the threshold for change occurs when salinity increases to between 1,000 and 2,000 mgL^{-1} (Brock *et al.*, 2005). High surface water salinity may influence emergence from the seed and egg bank in three ways; (i) reducing the viability of the dormant eggs and seeds, (ii) interference with germination/emergence cues, or (iii) toxicity effects kill any emergents / germinants (Nielsen *et al.*, 2003).

Increasing salinity may also have the potential to cause a shift between an aquatic plant dominated community to a phytoplankton and/or benthic mat dominated community (Davis *et al.*, 2003). For some species of freshwater aquatic plants, negative impacts on growth rates and development can occur above 1,500 μScm^{-1}, with the upper tolerance limit for most species approximately 6,000 μScm^{-1} (Nielsen *et al.*, 2003). All freshwater species will be lost when salinity exceeds 15,000 μScm^{-1}, with only salt tolerant species present (Nielsen and Brock, 2009). Loss of, or even shifts in the composition of aquatic macrophyte communities will affect the quality of habitat and food resources available for other biota (Nielsen and Brock, 2009). In a study on the foodwebs of the Coorong, Deegan *et al.* (2009) reported that under high salinity conditions, the decrease in potential food sources and biodiversity causes large bodied fish (bream, mulloway, flounder) to feed on sub–optimal food resources.

However, results from the Lower Lakes suggest that aquatic plants are more tolerant of high salinity in the lower reaches of the River Murray compared with areas upstream. Nicol and Ward (2010b) reported that there was no significant difference in the plant community that germinated from the seed bank of Lake Alexandrina when subjected to fresh (<1,000 μScm^{-1}) or surface water with a conductivity of 5,000 μScm^{-1}. Furthermore, they found that there was no lasting effect of surface water electrical conductivity up to 20,000 μScm^{-1} after sediments were exposed for six weeks (Nicol and Ward 2010b). In addition, Gehrig *et al.*

(2011a) reported *Typha domingensis*, *Phragmites australis, Vallisneria americana* and *Schoenoplectus validus* growing in Goolwa Channel (between Goolwa Barrage and Clayton) in areas where the surface water exceeded 20,000 µScm^{-1}.

4. Indigenous flows and environmental flows

The recently released Murray–Darling Basin Authority (MDBA) *Proposed Draft Plan: a revised draft* (2012: inside cover) states that more research is required to understand and finds ways to apply the concept of 'cultural flows' when considering Indigenous interests in water in the Murray–Darling Basin (MDB). The MDBA makes a clear statement of recognition at the front of the revised draft that includes the words of Ngarrindjeri elder Tom Trevorrow, Chairperson of South Australia's Ngarrindjeri Regional Authority (NRA):

> Our traditional management plan was don't be greedy, don't take any more than you need and respect everything around you. That's the management plan—it's such a simple management plan, but so hard for people to carry out.

In this chapter we provide a preliminary consideration of what we term Indigenous flows and make some preliminary comments about the relationship between this concept and environmental flows. Our discussion is written from a South Australian perspective, reflected in the words of Tom Trevorrow, and foregrounds the Indigenous philosophical understanding that the 'lands and waters are a living body' and that all things are a part of this living body (see Ngarrindjeri Nation 2006; Ngarrindjeri Nation 2012). Indigenous nations and their interests have only recently begun to receive some recognition from non–Indigenous governments and agencies across the MDB (see Morgan *et al.* 2006; MDBC 2006b; Hattam *et al.*2007; Hemming *et al.*2007; Weir 2009; MDBA 2012). This growing recognition has been slow and Indigenous community leaders are calling for recognition of Indigenous rights to water and to negotiate an Indigenous role in the management, enjoyment and use of waters across the MDB (see Hemming *et al.* 2011; Jackson 2012; MILDRIN 2012; NRA 2012; NBAN 2012; Weir 2012). The purpose of this chapter is to provide an introduction to Indigenous perspectives about water in the MDB and to discuss potential research and community engagement directions. We draw attention to recent successful South Australian examples of effective mechanisms and strategies for increasing Indigenous agency in natural resource management (see for example: Hemming *et al.* 2002; Morgan *et al.* 2004; Birckhead *et al.* 2011; Hemming *et al.*2011; Weir and Ross 2007; Weir 2009, 2012). Always included in Indigenous calls for a just recognition of rights is an appeal to non–Indigenous researchers and natural resource managers to shift their understandings of the relationship between humans and non–humans – conceived in Western terms as the divide between 'nature' and 'culture' (see Morgan *et al.* 2004; Latour 2005; Dhimurru 2006; Langton 2006; Ngarrindjeri Nation 2006; Hemming and Rigney 2008; Weir 2009; Birckhead *et al.* 2011; NRA 2012; MILDRIN 2012; NBAN 2012; FPWEC 2012). Also common to Indigenous perspectives on water is a call for recognition that Indigenous nations in Australia share much, but strongly retain the right to their own distinctive cultures, traditions and interests associated with water.

4.1. The political and historical contexts

MDB Indigenous nations share fundamental knowledges (epistemologies) and philosophies of being (ontologies) that connect them through their relationship to lands and waters (see Morgan *et al.* 2004; Weir 2009; MLDRIN 2012; NBAN 2012). At the centre of this relationship are the inland waters of the MDB. This culturally distinct relationship provides the platform for contemporary Indigenous partnerships in the region. The Basin has two Indigenous regional organisations recognised by the MDBA as critical to working with Indigenous nations – the Murray Lower Darling River Nations (MLDRIN) and the Northern Basin Nations (NBAN). These two collectives are comprised of representatives of individual nation groups. MLDRIN and NBAN have worked together on responses to the MDB planning process and both have made submissions on Indigenous interests in water including the issue of Indigenous 'cultural flows' (see MLDRIN 2012; NBAN 2012). These submissions address the requirements for further research and for new mechanisms for engaging Indigenous nations in water and natural resource planning. Social justice issues remain at the centre of Indigenous interests in water in Australia – Indigenous people in Australia were dispossessed of lands and waters without treaties.

In the MDB region in South Australia, native title and cultural heritage protection underpin Indigenous recognition and inclusion in Australian environmental management regimes. The MDBA's ongoing *Living Murray* initiative supports Indigenous engagement with NRM across the Basin and provides some financial support for Indigenous programs including the two representative bodies (see MDBC 2006). At a national level the MDBA and the National Water Commission (NWC) have sponsored Indigenous water research that includes the MDB region (see Toussaint *et al.* 2005; Toussaint 2009; Jackson *et al.* 2010; Jackson 2012; FPWEC 2012). The Northern Aboriginal Land and Sea Alliance (NAILSMA) is also actively pursuing a program of research, policy development and negotiation in relation to water across Australia's northern states (see Jackson 2007; Jackson and Morrison 2007; Morrison 2007; O'Donnell 2011).

In South Australia the First Peoples of the River Murray and Mallee (FPRMM) have a consent determination and Indigenous Land Use Agreement (ILUA) applying to lands and waters of the Upper Murray in South Australia (see Hemming *et al.* 2007; Mooney and Tan 2010). The Ngarrindjeri and Others Native Title Management Committee (NNTMC) is responsible for a native title claim that includes the River Murray to just north of Murray Bridge. The NRA is the peak regional Indigenous body that includes the NNTMC and includes the River Murray to Morgan through another member organisation – the Mannum Aboriginal Community Association Inc. (MACAI). Research has been conducted in this region by the Ngarrindjeri nation in partnership with Flinders and Charles Sturt Universities as part of a Water–For–A–Healthy–Country Flagship project (see Hemming and Rigney 2008; Birckhead *et al.* 2011). The South Australian Murray–Darling Basin Natural Resource Management Board (SAMDBNRMB) has also sponsored water–related research in the region, and is more recently working through a formal agreement with the NRA to progress water allocation planning (see Mooney and Tan 2010; KNY 2009).

4.2. Indigenous 'water' in the Murray–Darling Basin

Indigenous Australian philosophies of being are based on an interconnection between country, body and spirit. This interconnection is fundamental to wellbeing. As an example, the Ngarrindjeri nation in southern South Australia use the term *Ruwe/Ruwar* to encapsulate this concept and argue healthy lands and waters are critical to healthy Ngarrindjeri people and culture (see Bell 1998, 2008; Ngarrindjeri Nation 2006; Hemming *et al.* 2011). Ngarrindjeri have traditional responsibility to care for *Ruwe/Ruwar* and are engaging in negotiations and agreement making with South Australian authorities to transform the existing natural resource and heritage management regimes in the region towards recognition and support for healthy Ngarrindjeri *Ruwe/Ruwar*. The separation of this responsibility into compartments such as water allocation planning, community health, or wetland management planning is foreign to Indigenous philosophies of being and can be understood as a colonising strategy (see Hemming and Rigney 2008; Weir 2012). Indigenous caring for country ('natural resource management') always includes a taken–for–granted assumption of interconnection (see Ngarrindjeri Nation 2006; Goodall and Cadzow 2009; Weir 2009). Indigenous strategies and practices associated with 'water' planning and management necessarily take different forms to non–Indigenous approaches. In recognition of this complex set of differences, Indigenous leaders are calling for time, resources and opportunities to negotiate a new regime of NRM management across the MDB that includes a just recognition of Indigenous *a priori* rights. Indigenous leaders argue that research funding is required for Indigenous nations to conduct their own research projects to support their negotiations with non–Indigenous authorities (see MILDRIN 2012, NBAN 2012).

The MILDRIN (2012: 5) submission to the MDBA argues, '…that the *proposed Basin Plan* fails to articulate Sovereign First Nations and Traditional Owners rights to water'. We have used the term Indigenous flows to signal a distinction with the concept 'cultural flows' as it has been applied more broadly in water planning. Cultural flows are defined by MILDRIN (2012: 5) and NBAN as:

> Water entitlements that are legally and beneficially owned by the Indigenous Nations and are of a sufficient and adequate quantity and quality to improve the spiritual, cultural, environmental, social and economic conditions of those Indigenous Nations. This is our inherent right.

The NRA (2012: 3) makes the following point regarding *a priori* rights in its submission to the MDBA:

> Ngarrindjeri consider they have the first right, a right attached to the exercise of their cultural rights, interests and responsibilities, that precedes all other rights including but not limited to the legislative function of the MDBA to allocate water for particular uses. The rights and interests of the Ngarrindjeri require that water flows into, through, and from, their country from up river. This is a right a priori to all others and the MDBA should commence their consideration of allocations without interference or diminishment of these rights.

In 2009 the Ngarrindjeri nation in South Australia negotiated an agreement with the State of South Australia that recognised traditional ownership of Ngarrindjeri lands and waters and established a process for negotiating and supporting Ngarrindjeri rights and responsibilities for country (*Ruwe*) (see Hemming *et al.* 2011). The agreement takes the form of a whole–of–government contract agreement between the Ngarrindjeri nation and the State of South Australia. Called a *Kungun Ngarrindjeri Yunnan* agreement (KNY – Listen to what Ngarrindjeri have to say), it provides for a resourced, formal structure for meetings and negotiations between the Ngarrindjeri Regional Authority (NRA – peak body) and government, universities and other non–Indigenous organisations (see Ngarrindjeri Nation 2006). The NRA includes in its vision for Ngarrindjeri people, the following overarching statement:

> Our Lands, Our Waters, Our People, All Living Things are connected. We implore people to respect our Ruwe (Country) as it was created in the Kaldowinyeri (the Creation). We long for sparkling, clean waters, healthy land and people and all living things. We long for the Yarluwar–Ruwe (Sea Country) of our ancestors. Our vision is all people Caring, Sharing, Knowing and Respecting the lands, the waters and all living things.

(Ngarrindjeri Nation 2006:5)

This vision makes clear the essential link between the wellbeing of Ngarrindjeri individuals, families, communities and the interconnection to lands and waters. This Ngarrindjeri worldview is gaining high–level acceptance in the non–Indigenous context through official State Government recognition of the *Ngarrindjeri Nation Yarluwar–Ruwe Plan* (2006) and KNY agreements. In recent years the Ngarrindjeri KNY strategy has provided the platform for South Australian government support for building Ngarrindjeri capacity to take a leading role in caring for Ngarrindjeri country – including conducting research to better inform Ngarrindjeri programs (see DEH 2009). This developing Indigenous capacity provides an opportunity for collaborative research aimed at supporting new strategies for NRM and water management in the MDB. Recital E (KNY 2009 in Hemming *et al.* 2011: 110) provides an indication of the intentions of the agreement:

> E. By this Agreement the Ministers wish to provide support and resources to the Ngarrindjeri Regional Authority Inc. and enter into negotiations and consultations with the Ngarrindjeri about

> the maintenance and protection of Ngarrindjeri culture and cultural sites and the natural resources of the Land [lands and waters].

4.3. Conclusions

With formal agreements, careful planning and funded programs, the Indigenous nations in the MDB are pursuing a strategic approach to securing improved wellbeing for community, family , individuals and lands and waters (see Ngarrindjeri Nation 2006; Hemming *et al.* 2011; FPWEC 2012; Jackson 2012). Environmental research and NRM are an important part of the broad Indigenous program in the MDB and they bring opportunities for supporting Indigenous agency in the development of regional Australian futures (see Hemming and Rigney 2008). In New Zealand Maori have negotiated a co–management agreement over the Waikato River that incorporates a process of assessment and monitoring of the cultural health and wellbeing of the River. Innovations such as this can provide a pathway for understanding the relationship between Indigenous conceptualisations of water, Indigenous flows and water planning tools such as environmental flows. Recent negotiations and project development between the Ngarrindjeri Nation, the Department of Environment and Heritage (DEH) and the Department for Water (DFW), have started a comparative process for research, planning and management of the Coorong, Lakes and Lower River Murray Region in South Australia (DEH 2009; Hemming *et al.* 2011).

5. Predictive ecology

With the global decline in water–dependent ecosystems as a result of human development, environmental flow frameworks have evolved from basic flow threshold analyses, to consideration of flow requirements for complex ecosystems. The review below introduces frameworks and assessment methods, with a focus on alternative approaches for assessing the outcomes of flow regimes. The second part introduces modelling approaches, with a focus on approaches that are typically used in a decision–making environment. Modelling approaches are focused on those that can sit within a decision–support environment.

5.1. Frameworks and Assessment Methods for Environmental Flows

Rivers are extremely complex systems characterised by a range of interactions between geomorphology, climate, hydrology, and ecological processes. There are many approaches to determining the environmental water needs of lowland river ecosystems where changes in flow habitat and physical habitat are an important determinant of the distribution, abundance and diversity of rivers and streams (see reviews by Arthington and Pusey 1993; Tharme 2003; Arthington and Zalucki 1998). Numerous ecologically relevant flow metrics and techniques have been derived for quantifying human–caused flow and ecological alteration (for example Arthington *et al.* 2006; Arthington *et al.* 2007; Poff *et al.* 1997; Puckridge *et al.* 1998; Richter *et al.* 1998; Richter *et al.* 1996). Tharme (2003) reported that there are in excess of 200 individual methodologies within four main classifications that have been used to determine environmental water requirements. Typical classifications used are:

- hydrological
- hydraulic rating (habitat retention)
- habitat simulation
- holistic methodologies

To varying extents, the first three approaches assume that a volume of water will provide the drivers required to achieve management objectives. However, it is critical to acknowledge that water is only one important aspect for maintaining ecosystem function (Arthington *et al.* 2010); the provision of water is a critical link in the ecology of wetland and floodplain systems but that does not automatically imply that the link is functional (Jenkins and Boulton 2003). It is not just hydrology or geological processes that engineer landscapes. Organisms can change the physical structure of landscapes, creating, modifying and maintaining species habitat and modulating the flow of resources (Dollar *et al.* 2007), with extreme events (floods and droughts) representing key selective processes driving mortality and recruitment (Lytle and Poff 2004). It is also unclear what the ecological ramifications of only partially meeting a specific EWR are.

In this report we review a broader set of approaches for predicting ecological outcomes of flow regimes. Ecological response model types reviewed here include:

1. Hydrologic alteration
2. Habitat suitability modelling
3. Population modelling of select ecological outcomes
4. Biodiversity (hydrological diversity)

5.2. Hydrological Alteration (Environmental Water Requirements)

Hydrological methods rely primarily on flow measures and indices, which are drawn from a reference hydrological time series data (e.g. natural flows or modelled surrogates), most commonly represented as daily mean flow. These methods are regarded as the simplest approach to determining environmental flows and primarily rely on the statistical analysis of the time series of flow (Tharme 2003).

Early methods for assessing hydrological change focussed in setting minimum flows, with the process for establishing these being regarded as mostly arbitrary. It is now widely accepted that minimum flows are inadequate for maintaining the structure and function of a riverine ecosystem (Arthington *et al.* 2006). A more sophisticated method of environmental flow assessment was developed by Tennant (1976). This method uses summary tables of flow statistics to assess flow requirements for maintaining fish habitat (Tennant 1976). This method is still used selectively today. As time has progressed, the contemporary dominant paradigm in environmental flow assessment has emerged as the natural flow paradigm (Poff *et al.* 1997; Richter *et al.* 1997).

The theory of the natural flow paradigm states that to protect freshwater biodiversity and maintain the essential goods and services provided by rivers, we need to mimic components of natural flow variability (Poff *et al.* 1997; Arthington *et al.* 2006; Richter *et al.* 1997). To do this, the magnitude, frequency, timing, duration, rate of change and predictability of flows (e.g., floods and droughts), and the sequencing of such conditions needs to be restored to rivers and floodplains (Arthington *et al.* 2006; Poff *et al.* 1997). Whilst long term changes in flow magnitude, duration, timing and frequency is believed to define physical habitat over sub–catchment to catchment spatial scales, short–term histories of hydrologic events are believed to influence habitats at reach to within–reach scales (Kennard *et al.* 2007).

Hydrological methods of assessment frequently use these metrics of magnitude, frequency, timing, duration, rate of change and sequencing (predictability) of events for environmental flow assessments. The important parts of the hydrograph for assessing these metrics have been further defined in (DNRE 2002) as being:

- Cease to flow: No discernible flow in a river, which may lead to either total or partial drying of the river channel.
- Low flow: Low flows provide a continuous flow through the channel, providing flow connectivity between habitats in the channel.
- Freshes: A fresh is a small and short duration of peak flow, which can exceed the base flow and last for several days.
- High flows: High flow is a persistent increase in the seasonal base flow that remains in the channel.
- Bankfull flows: Flows are of sufficient magnitude to reach bankfull condition, with little to no flow spilling onto the floodplain.
- Overbank flows: Flows result in the inundation of the adjacent floodplain.

An alternative to hydrological methods is hydraulic rating methods, which consider the channel morphology of individual rivers. These methods use a discharge–wetted perimeter plot. As with early hydrological approaches, habitat preferences are often defined on a subjective basis. They also require river channel assessments, which is an impractical exercise at a large spatial scale. Furthermore, like hydrological methods, hydraulic rating methods do not directly take into account the habitat requirements of a target species (Gippel and Stewardson 1998).

Specific criticisms of EWRs are their limited explanatory power in modelling transitional changes of ecosystems. In analysing an EWR, it is only possible to determine whether an EWR is met as hydrologically specified, without accounting for temporal (population, structural) and spatial (area) representations of change. This limits the analysis of the ‘so–what’ elements. EWRs are also limited to hydrologic drivers, failing to recognize the other non–flow drivers that impact ecological outcomes. Being an asset–based approach also fails to acknowledge the importance of landscapes in maintaining resilience, a diversity of habitat and functional processes at nested scales.

Recent advances

Extensions to hydrological assessment approaches include the Sustainable Boundary Approach (Richter 2010) and the Ecological Limits of Hydrological Alteration (ELOHA) (Poff *et al.* 2010). Both approaches are similar in focus, where “human–induced alterations of water flows and water quality within ‘sustainability boundaries” (Richter 2010), they represent an application of the natural flow regime concept (Poff *et al.* 1997; Richter *et al.* 1997) and have a focus on flow change at regional scales. The Sustainability Boundary Approach aims to set limits on the extent of hydrological change as a consequence of water withdrawals and discharges, water infrastructure operations and land uses can alter natural variability (Figure 5.1). The proposed outcome is maintenance of the dynamism of river systems within an adaptive framework. Likewise, ELOHA seeks to provide a framework for assessing ecological consequences of different ‘levels’ of flow modifications, where analyses of flow alteration are used to develop a classification of river types (Poff *et al.* 2010). Both methods advocate the development of flow–ecological responses, where ecological changes are expressed as percentage ecological change given a percentage in flow alteration (Arthington *et al.* 2006; Poff *et al.* 2010). Both are participatory values–driven process.

(Arthington *et al.* 2006; Poff *et al.* 2010; Richter 2010) acknowledge the challenges in deriving flow–ecology relationships to inform the flow management process. The rapid acceptance of the natural flow regime concept has been accompanied by an expectation that ecologists can easily provide specific environmental flow prescriptions for riverine ecosystems (Poff *et al.* 2010). Unfortunately, translating general hydrologic–ecological principles and knowledge into specific management rules for particular river basins and reaches remains a daunting challenge (Poff *et al.* 2010). Indeed, our ability to model hierarchical linkages across water dependent ecosystems remains largely unrealised, with our understanding of linkages being poorly understood, inadequately validated, and largely expressed as hypotheses or conceptual representations, rather than predictive or quantitative models (Kennard *et al.* 2007).

Other criticisms include the lack of representation of interactions between species, life stages and other variables that influence the state of the ecosystem and index variable (Ayllón *et al.* 2011) , they do not deliver information on demographic rates (Lancaster *et al* 2010) and they cannot deliver information on the population viability of species (Shenton *et al.* 2010). Such assumptions limit advances in understanding of stream ecosystems and how to manage flows, and undermine the these model forms from having an representation of risks associated with different flow scenarios

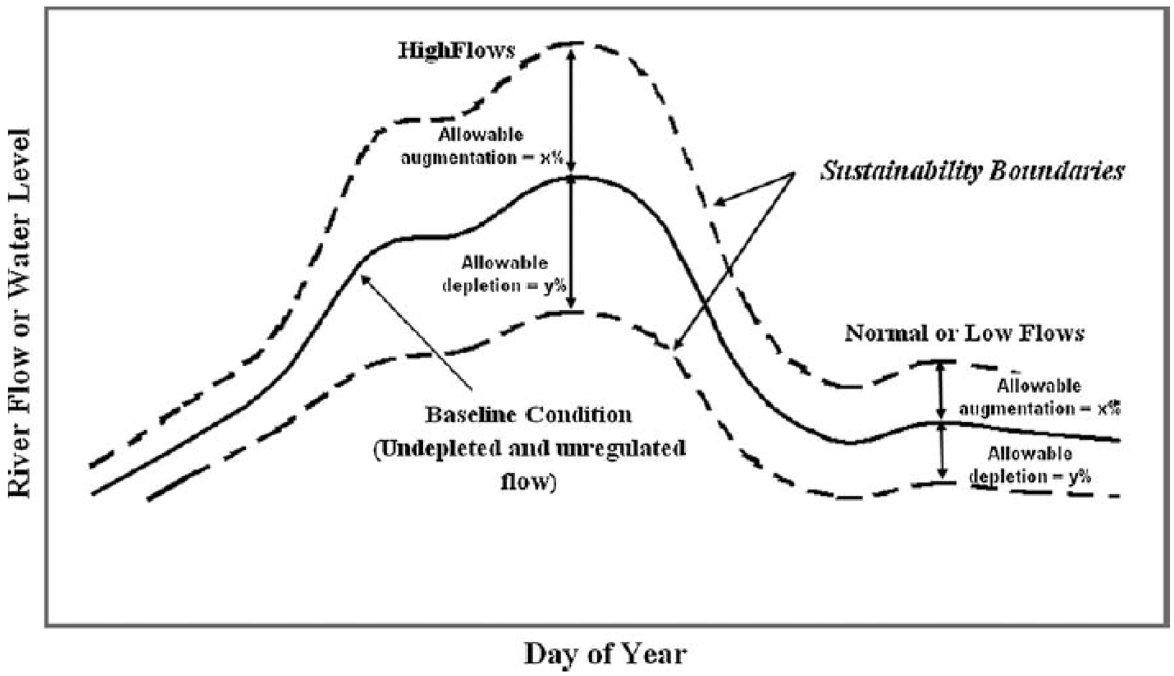

Figure 5.1: Sustainability Boundary Approach to environmental flows. Source: Richter (2010).

Connectivity

One of the limitations in hydrological indices is the limited representation of connectivity. Understanding changes in lateral and longitudinal interactions between the channel and the floodplain are critical for understanding the biota of natural systems and the changes caused by alterations induced by river regulation. Flow patterns and connectivity between river channels, wetlands, floodplains and groundwater system is essential to the viability of many water dependent species (Bunn and Arthington 2002), leading to characteristic high biodiversity (Ward *et al.* 1999). Maintenance flows for channel form provides for habitat complexity, native biota have evolved life histories specific to flow regimes where the provision of flows act as spawning triggers and ensure recruitment success, lateral and longitudinal connectivity provides ecological connectivity across landscapes and between habitat patches, and the flow regime established favours niche habitats for native species, discouraging invasions by alien species (Bunn and Arthington 2002). Ecological connectivity provides for biodiversity at many levels of biological organization, from genes to landscapes, at a variety of spatial or geographic scales (Ward and Stanford 1995; Poiani *et al.* 2000).

The role of lateral connectivity in connecting rivers and floodplains promotes migration, improved condition of fish species, acts as a reproductive cue for select fish species and vegetation types, promotes local and large–scale migrations of species, acts to modify channels promoting heterogeneity in habitats, increases productivity through the release of organic matter and nutrients and promotes biodiversity (Ward and Stanford 1995).

In the southern Murray Darling Basin, the relationship between connectivity of wetlands and the diversity of water dependent vegetation species was explored, and a clear increase in the probability of presence and species diversity was shown (Figure 5.2) (Ganf *et al.* 2010). Connectivity was expressed as the number of days per year a wetland receives water from a river. Species diversity was modelled using field data

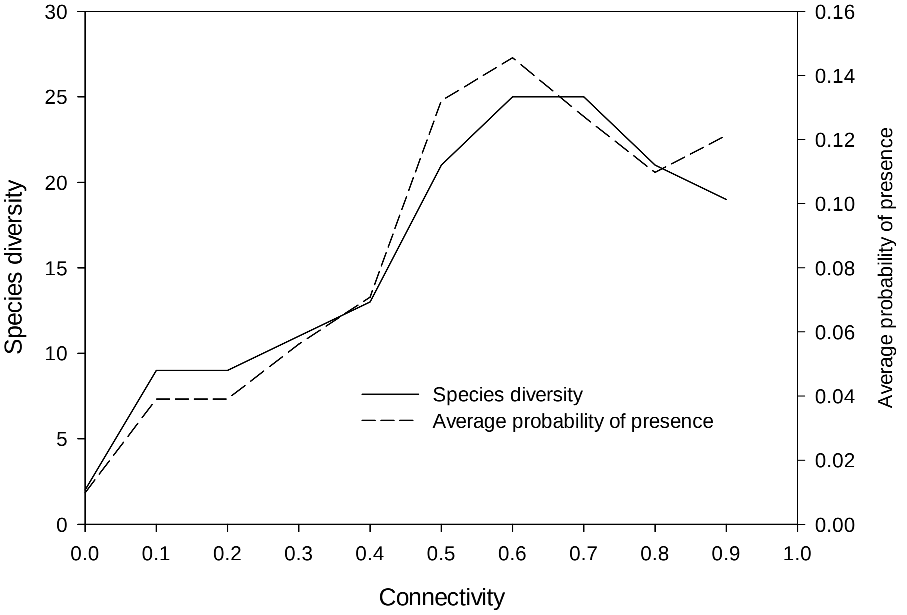

Figure 5.2 Relationship between diversity of water–dependent vegetation as a function of connectivity in the southern Murray Darling Basin Source: Ganf *et al.* (2010).

5.3. Habitat Suitability models

Habitat suitability models are generally static representations of change, which are often represented spatially using probabilities or preferences. They are used to: to predict species occurrences on the basis of abiotic and biotic variables; to improve the understanding of species–habitat relationships; and to quantify habitat requirements (Ahmadi–Nedushan *et al.* 2006). They focus on the feature of a habitat that influence habitat selection and enhance fitness, where habitat preferences are used to reflect the integration of multiple environmental factors across multiple spatial scales (Chalfoun and Martin 2007).

Habitat suitability is often expressed as a habitat suitability curve (e.g. suitability index curve, probability of use or preference curve), often seasonally defined, which specifies the assumed seasonal requirements of different species, life stages or habitat guilds (Lamouroux *et al.* 1998; Lamouroux *et al.* 1999; Welcomme *et al.* 2006). Curves are used to depict the relationship of a target organism's response to a gradual changing habitat variable scaling from unsuitable to suitable (often expressed as an index of 0 to 1). Three different types of habitat suitability indices have been distinguished by Bovee (1986) as referenced in (Jowett *et al.* 2005): Category I: professional judgement indices; Category II: habitat use indices; Category III: habitat preference indices. Category I indices are derived from life history studies in the literature, or professional judgement. Category II indices use data collected specifically for habitat studies, based on frequency of occurrence of actual habitat conditions used by different species and life stages in a stream. Category III data combine a category II frequency analysis with additional information on the availability of habitat combinations in the sampling reaches. Suitability indices can be created separately for different habitat variables (e.g. cover, substratum size, low and high flows) and these indices are then combined to define a composite suitability index. Different methods have been used to combine the different indices into a single outcome.

A broad range of modelling tools has been developed to predict how physical habitat conditions (i.e. water depth, velocity, cover, substrata) change with discharge. The Instream Flow Incremental Methodology

(Bovee 1982) and its associated software the Physical Habitat Simulation System (PHABSIM) (Shuler and Nehring 1993), are widely used habitat suitability assessment tools. Typically, these require hydrological and hydraulic data for a grid of cells in a river stretch are compared with the habitat suitability of a target species. The Murray Flow Assessment tool (MFAT) (Young *et al.* 2000) uses a similar modelling approach to PHABSIM, where habitat requirements are modelled as preference curves. MFAT considered a range of water–dependent taxa, including fish, vegetation and waterbirds, with species represented within into functional groupings. Available literature and expert opinion was used to derive a range of response curves for relevant flow–related habitat conditions. Response curves were developed for variables such as flow and spawning timing and flow duration for fish, and were spatially differentiated where considered appropriate. More recent incarnations of the approach can be found in EcoModeller (Marsh *et al.* 2007) and in the IBIS DSS (Merritt *et al.* 2009; Merritt *et al.* 2010b). These models use habitat suitability indices that can be classed as a Category I, using the classification process of Bovee (1986).

In habitat suitability models, preference curves for individual flow attributes (most commonly, frequency, duration, timing, rate of rise and fall) are combined as a composite score. In EcoModeller, the most limiting factor is used (Marsh *et al.* 2007), whereas IBIS allows comparison of an arithmetic mean and an expert–driven weighting process (Merritt *et al.* 2010b). MFAT uses a range of combination methods, which were derived from expert judgements made by the Regional Evaluation Groups (REGs) (Young *et al.* 2003). There is no standard method available to guide this process, and the process is largely subjective.

Criticisms of this approach are that the outcomes of models consider single events in isolation, are based on limited data or poorly documented thresholds (Miao *et al.* 2009) or are subject to the fallacies associated with expert opinion (Cooke 1991). Moreover, these approaches generally implicitly ignore potential responses of multiple extreme events, and the connectivity of events (Miao *et al.* 2009). Other criticisms include the lack of representation of interactions between species, life stages and other variables that influence the state of the ecosystem and index variable (Ayllón *et al.* 2011).

Population modelling

Population models are often highly deterministic in nature, where for example, an estimate of the changes in the number of a species is predicted for a given change in habitat. These models generally require an extensive knowledge of a species life history, and need a full representation of the variables at a site that can influence populations of individuals.

In modelling individual species, knowledge of its distribution across the landscape, and how it performs at the population level (e.g. recruitment, overall fitness) is required. Knowledge about structure (age– or size–class distributions) of a population is helpful in determining what life–stage is most influenced by flow regime changes. Age or size distributions provide evidence of successful recruitment and functional assessments of life–stages can be used to yield significant insight into the structure of the population. Age–class distributions may also provide insight into event–driven recruitment that may be statistically related to hydrologic characteristics (Merritt *et al.* 2010a). Whilst models of birds can be limited to analyses of age structure (Arthur 2011), vegetation models require size and stage structured models, particularly as individuals can vary in size by five orders of magnitude (Loehle 2004).

Population modelling can be used to predict how different flow regimes in different years could provide for the requirements of a range of species, with a focus on the temporal responses of different species to changes in flow regime varies widely. Short–lived species such as annuals could respond on the scale of a single season, but longer–lived species could require decades or even centuries to respond. A variety of stochastic biotic and physical factors (fire, herbivory, competition, disease, fluvial disturbance) could also influence the distributions of species and if not explicitly incorporated into the model, these factors increase the uncertainty of predictions (Auble *et al.* 2005).

Metapopulations are a concept that has emerged more recently, where sub–populations are linked through individuals and gene flows, and are characterized as having sources and sinks. Metapopulation structure and processes can be regarded as essential for sustaining local–scale species and small patch ecosystems, where replication and connectiveness is more important than size (Loehle 2004). A recent example of metapopulation modelling has focused on waterbirds (Arthur *et al.* 2012; Arthur 2011). Waterbirds breed throughout the Murray–Darling Basin and Australia, sometimes in mixed colonies numbering in the hundreds of thousands (Marchant and Higgins 1990). Waterbirds typically breed after extended periods of flooding, first to initiate breeding and then allow completion of breeding through to successful fledging (Briggs and Thornton 1999; Reid *et al.* 2009). If thresholds are not exceeded, breeding will not occur. Once exceeded, the size of the breeding event (i.e. number of nests) can be positively correlated with the area of floodplain inundated (Kingsford and Auld 2005; Kingsford and Johnson 1998). Population modelling of waterbirds suggests that sustaining populations of egrets will require breeding of most of the egret population every 1 – 2 years (Arthur in press).

Over the long term, developing the linkage between habitat simulation models and dynamic population models is essential to improve the prediction of population changes with habitat.

Biodiversity Models using Systematic Conservation Planning

Species distribution models are numerical tools that use data on species occurrence or abundance estimates (Elith and Leathwick 2009). Species distribution models typically assume that habitats are closed, stable and without competition (Cassini 2011). Most species distribution models produce a habitat suitability map as an output. Models include Generalised Linear Models, Generalised Additive Models and Artificial Neural Networks. These models are empirical, fitting relationships between habitat variables and species presence data.

A common framework used for conservation of biodiversity assessments is species conservation planning. Although applications have typically focused on terrestrial systems, there are a growing number of applications in freshwater environments (Linke *et al.* 2011; Hermoso *et al.* 2011; Streckler *et al.* 2011; Nel *et al.* 2011). Species conservation planning seeks to develop a plan which ensures the persistence of biodiversity. Inputs to a systematic conservation model include species presence, modelled species surrogates or physical surrogates (Linke *et al.* 2011). In systematic conservation planning, the key steps include: 1. collate data on biodiversity features and set quantitative targets to a desired conservation outcomes; 2. list the management options, their costs and the impacts on stakeholders; 3. set constraints (maximum cost or impact on stakeholders); 4. maximising the outcomes in (1) under these constraints (Margules and Pressey 2000).

5.4. Environmental models for decision–making purposes

After selecting the ecological scope and conceptual framework for a model, the next step is in specifying an appropriate style of representation for your model. The review below provides an overview of different model forms and their functionality.

A model assists in the decision making process by assembling the many different interacting factors that need to be considered in understanding a problem and investigating alternative and feasible solutions. By their very design, models are incomplete representations of managed systems (Walters 1997). A model is a simplified representation of a real system, containing the characteristic features that are essential in the context of the problem to be solved or described (Jorgensen and Bendoricchio 2001). The ever growing knowledge base challenges us to develop more efficient ways of organising, processing, and analysing ecological knowledge to emphasize and facilitate the process of ecological reasoning (Rykiel 1989).

Ecological modelling approaches span a range of complexity, ranging from conceptual models to complex dynamic simulation models, and have been categorized using various typologies. Models can be classified according to their design and representation of a system. Distinguishing model characteristics can take into consideration its use as well as features of its construction and system representation. Some such distinguishing characteristics of models are: scientific or management; stochastic or deterministic; compartment or matrix; reductionistic or holistic; dynamic or static; distributed or lumped; and causal or black box. The choice of model approach is highly dependent on the question being asked.

A distributed model is usually based on ordinary differential equations. An example of a distributed model is advection–diffusion model for transport of a dissolved substance along a stream (Jorgensen and Benderoicchio 2001). A dynamic model uses a differential (continuous) equations or difference (discrete) equations to describe time (Jorgensen and Benderoicchio 2001). Dynamic model are typically deterministic model which is computationally intensive that represent dynamic feedbacks, and in ecology, is typically subject to assumptions. An example of a dynamic model is population dynamic models, which can incorporate extinctions, invasions, succession, and predator–prey cycles. A stochastic model contains stochastic input disturbances and random measurement errors (Jorgensen and Benderoicchio 2001). This type of model can be reduced to a deterministic model if the input disturbances and random measurement errors are assumed to be zero. A deterministic model assumes that the future response of the system is completely determined by knowledge of the present state and future measure inputs.” (Jorgensen and Benderoicchio 2001).

Black box models looks at how outputs are affected by changes in inputs. They only looks at what is measureable and they link inputs to outputs, without expressing causality or logic in the internal mechanisms. Causal models on the other hand are internally descriptive, enabling examination of how the inputs, states and outputs are all connected to, and affect, each other. Causal models are more widely used in environmental sciences than the black–box model, due mainly to the improved understanding that a causal model gives the user the function of the system including the many chemical, physical and biological functions (Jorgensen and Benderoicchio 2001). An example of a causal model is a Bayesian network. The following sections describe the types of model approaches commonly used analysing environmental problems, within a decision–context.

Common model types used to support environmental decision making

A range of model types can be applied for analysing issues, within a decision environment. Here we consider models that are typically used to combine or integrate multiple system processes (e.g. time series of flow to ecological response), within a management context. This section reviews approaches used most commonly in environmental decision making, and it is not an exhaustive review. Whilst any of these model types can be applied, the selection of the approach will be guided by a set of use criteria, as shown in Table 5.1.

Table 5.1: A list of model types, showing their major features and methods for handling model uncertainty. Adapted from Jakeman *et al.* (2007)

Use Criteria		**Process Models**	**System Dynamics**	**Decision Trees**	**Bayesian networks**	**Agent–based**	**Hybrid**
Purpose	Prediction	X		X	X		X
	Explorative	X	X	X	X	X	X
Input data	Qualitative			X	X		X
	Quantitative	X	X	X	X	X	X
	Both				X		X
Breadth of analysis	Focused, in depth	X	X		X	X	X
	General, Broad		X	X	X	X	X
	Either		X		X	X	X
Space and time	Spatial		X	X	X	X	X
	Dynamic	X	X			X	X
Uncertainty	Method applied	Monte Carlo, Sensitivity	Monte Carlo, Scenario analysis	Sensitivity, Scenario analysis	Probabilities, Sensitivity, Scenario analysis	Sensitivity	Probabilities, Sensitivity, Scenario analysis

Below is a brief review of each of these modelling approaches. This review builds on material presented in (Jakeman *et al.* 2007). Modelling approaches focus on those that can sit within a decision–support environment, rather than statistical approaches, which are more appropriate for analysing empirical relationships.

Process Models

Process models are typically made up of a set of physical equations, which are generally described deterministically. In ecology, the types of process models can include a biodemographic, bioenergetic and biogeochemical models. A biodemographic model can describe the number of individuals, species or classes of species for modelling of life cycles (Jorgensen and Benderoicchio 2001). A bioenergetic model typically describes energy flows, and therefore the units of the state variables will be units of energy (kW, or kW per unit area or volume). A biogeochemical model describes the flow of material, and therefore the units of the state variables will be units of mass (kg or kg per unit area or volume).

Standard equations are used in model development, and include equations of conservation of mass, momentum and energy (Jorgensen and Benderoicchio 2001). Process based models typically have a large quantity of spatially distributed parameters. Limitations of process models include (Letcher and Jakeman 2004),

- Model parameters often have to be calibrated against unobserved data (due to the large number of parameters, the fact that system features are often unobservable and/or inaccessible and the heterogeneity of important features within the system).
- Equations often require continuous data (temporal and spatial) but in models the data is often point source data, which is assumed to be representative of an entire area

- Error accumulation can occur through this summing of small scale processes to represent larger scale processes.

- Because of the large amount of spatially and temporally distributed data required in process based models, they can be very costly and demand intensive both in terms of computer requirements and expertise of the modellers.

System Dynamic Models

System dynamic models are underpinned by systems theory, a method used to enhance the understanding of the dynamic behaviour of complex systems. They are generally used to model dynamic systems, representing internal feedbacks and time delays. The model architecture used loops and stocks and flows and can be used to represent systems that are nonlinear. In practice, system dynamics models are made up of a set of coupled differential equations. However, system dynamic models are difficult to parameterize, relying on a complete knowledge of system relationships.

System Dynamics models are applied in theoretical ecology, as well as other sciences, where models are predominantly theoretical. It is difficult to build these models using data, as rarely is the data available to represent such dynamic processes. They are difficult to validate and thus are not particularly useful for analysing the uncertainties in an ecological system. Given their theoretical nature they have limited predictive capacity. A common criticism of the approach is models can be viewed as ‘black boxes’, but this criticism can be made about many modelling approaches.

System dynamics are often used for scenario analysis of alternative policies, for social learning to improve the understanding of a system or to compare the sensitivity of alternative system assumptions (Jakeman *et al.* 2007). The strength of the approach is in the ability to represent complex links in the human–biophysical environment, including spatial or temporal lags, nonlinearities and feedback loops (Costanza and Ruth 1998).

Decision Trees

A decision tree tool shows the different paths available in analysing decision problem. These pathways represent different decision alternatives. The possible scenarios and decisions are represented as linked branches from left to right showing the order in which they would occur (Gedig 2006).

Decision trees provide a highly effective structure to capture and explore decision options and investigate the potential outcomes of these options. They can help in balancing risks and rewards associated with each possible course of action. A decision tree is used in data mining and machine learning as a predictive model; that is, a mapping from observations about an item to conclusions about its target value. Decision trees are simple to understand and interpret; require little data and can be combined with other decision techniques. In a study by (Failing *et al.* 2007), a decision tree was used to evaluate the costs associated with water releases from a dam and the benefits of these releases to downstream salmonid populations in Canada. The model was used directly in stakeholder negotiations.

Limitations of the technique are the simplistic representation of ecosystems, they can be highly subjective and do not handle feedbacks or represent system dynamics.

Bayesian Networks

Bayesian networks are graphical models that represent variables as part of a causal chain. The strength of linkages between cause and effect are described probabilistically. In assessing the outcomes of environmental flows, Bayesian networks can be used to link flow scenarios with ecological outcomes (Merritt *et al.* 2010b; Pollino *et al.* 2007).

Depending on the level of knowledge, variables can be expressed qualitatively (i.e. in a categorical form such as low, medium, high) or quantitatively or a combination of the two. Probabilities are used to describe how variables combine. These probabilities can be estimated using expert. BNs are often used for analysing and communicating causal assumptions not easily expressed using mathematical notation (Pearl 2000). They are particularly useful for uncertainty analysis as they have the ability to consider inadequate knowledge or understanding of system processes, inherent randomness, subjective judgement and vagueness in parameter estimation, disagreement, measurement error and sampling error (Morgan and Henrion 1990).

Bayesian networks are increasingly being applied by the wider modelling and decision–theory community for addressing a diversity of environmental issues. Their appeal lies in their ability to:

- be used in scenario analysis, directly linking decisions to outcomes;
- represent the uncertainty or risk associated with decisions and outcomes;
- be used as a framework for integration, where integration can be across disciplines and across data sources (qualitative and quantitative);
- incorporate analysis tools, for example to test the sensitivity of decisions to desired outcomes or perform multi–criteria analysis of outcomes;
- easily allow updating of the knowledge and data embedded in them;
- be used in participatory contexts.

Limitations of Bayesian networks include their inability to incorporate feedback loops and their poor representation of spatial and temporal dynamics. To overcome these limitations, Bayesian networks can be used as part of a hybrid modelling approach, where different types of models are linked within a hierarchical system. Such an approach was used in developing an environmental flow model and tool for NSW inland wetlands (Merritt *et al.* 2010b).

Agent–based Models

Agent based models are a form of coupled complex models that explore the interactions between individuals in a system, referred to as agents. Agents are components of software that: contain data and code; adapt to changes in their environment; can inhabit unpredictable and dynamic environments; are rational and enable problem solving; are effective in managing systems that are, open, unpredictable, complex and large (Smith and Mackaness 1996). An agent can be a human or an alternative biological entity.

Given a network of agents which share information and interact with each other, the situation is termed a multi–agent system (Costanza 1998). Agents represent an interface between the problem, the decision environment, and the model user (Chigmell and Hancock 1994). They are a solution, whereby a non–expert user can navigate both the data sources and structure of the problem.

Agent based models are able to consider a range of important factors of the focus system. They are also very flexible, allowing further information and simulations to be incorporated later without affecting the existing interactive components. However, agent based models are generally not as useful for prediction as coupled complex models or Bayesian networks as they are generally hypothetical (Jakeman *et al.* 2007).

What they are useful for is exploring the possible outcomes of various management options through their representation of complex systems, often involving many stakeholders who may have competing interests.

Hybrid models

Hybrid models are a combination of models, and can be composed on any set of interacting models, including those described above. The overall model or system is made up of many component models, and these models may be of different types. For instance they may include expert models (i.e. decision trees, linguistic, rule–based), BNs, computational models, metamodels or statistical or empirical models). Due to their hybrid nature they are flexible, however, with that characteristic comes a requirement for a broad base of technical competence and there is limited availability of technical platforms.

Because they can incorporate a diversity of knowledge, hybrid models encourage participation, including members of the public, government, interest groups, stakeholders and business groups. This process, as well as being an opportunity to elicit well rounded knowledge and encourage participation and ownership of the problem, also provides transparency in the modelling process.

An example of a hybrid modelling system is the models that underlie the IBIS Decision Support System (Merritt *et al.* 2010b). The models combine outputs from a process–based hydrologic model (IQQM) as inputs to Bayesian network ecological response models. The Bayesian networks are dynamic within the modelling platform, with the outcome of one event informing the next. Models also incorporate uncertainties in predictions. Ecological models have been build for vegetation and waterbird species in three NSW wetland systems.

6. Options for developing an environmental flow tool

A move to holistic models that assess the EWR's of the entire riverine ecosystem with explicit links to the full range of the hydrological regime is underway at the international scale (Tharme 2003). To be fit for purpose, the model that will be developed for use is likely to needs to incorporate bottom up, and top–down drivers. Management solutions often attempt to combine disjointed inputs from multiple disciplines rather than utilising an interdisciplinary approach. Interdisciplinary science is needed to converge separate disciplines at a common understanding of the issues, relevant temporal and spatial scales, underlying processes, variables and indicators (Dollar *et al.* 2007), interactions and feedback mechanisms. Conceptual models that focus on specific aspects, i.e. role of sources of carbon, fish, birds, geomorphology are particularly useful, but are typically isolationist in their ability to predict outcomes from a management action on a landscape scale.

A draft conceptual model for outcomes of enhanced low variability in the lower River Murray is provided in Figure 6.1. Although the model is simplistic, it depicts interactions between multiple processes. To be truly effective, the model need to include the (i) driver of abiotic and/or biotic change, (ii) the substrate which is acted upon by the driver, (iii) controllers of the driver; and (iv) the entity or process that responds. It is important to understand, and subsequently specify scale of interactions. This is because a responder at one scale is likely to be a driver at another (Dollar *et al.* 2007) leading to a series of feedback mechanisms and interdependencies.

6.1. Management objectives and ecological targets

The 'management objectives' that have been established for key sites within the Murray–Darling Basin are typically value judgements rather than 'targets' based on a robust understanding of ecological thresholds or tipping points. To date, few sites within the South Australian section of MDB have established 'Ecological Targets'. Quantifiable and testable targets allow for an unambiguous determination as to whether a pre–determined level of condition has been achieved. However, setting targets is a task that must be approached with due caution for the following reasons:

1. in many cases sufficient baseline data and understanding of ecosystem function does not exist to set targets that may be ecologically meaningful
2. targets that are not based on verified ecological interactions are value judgements that may result in outcomes that are detrimental to other components of the ecosystem
3. the use of tightly defined ranges of a variable that is suitable for a single species or groups of species ignore interactions
4. managers may drive all components of complex systems towards the same endpoint, reducing the variability and habitat mosaic which characterise floodplains and underpins diversity
5. a target specifying an ecological or biological endpoint overlooks the concept of resilience

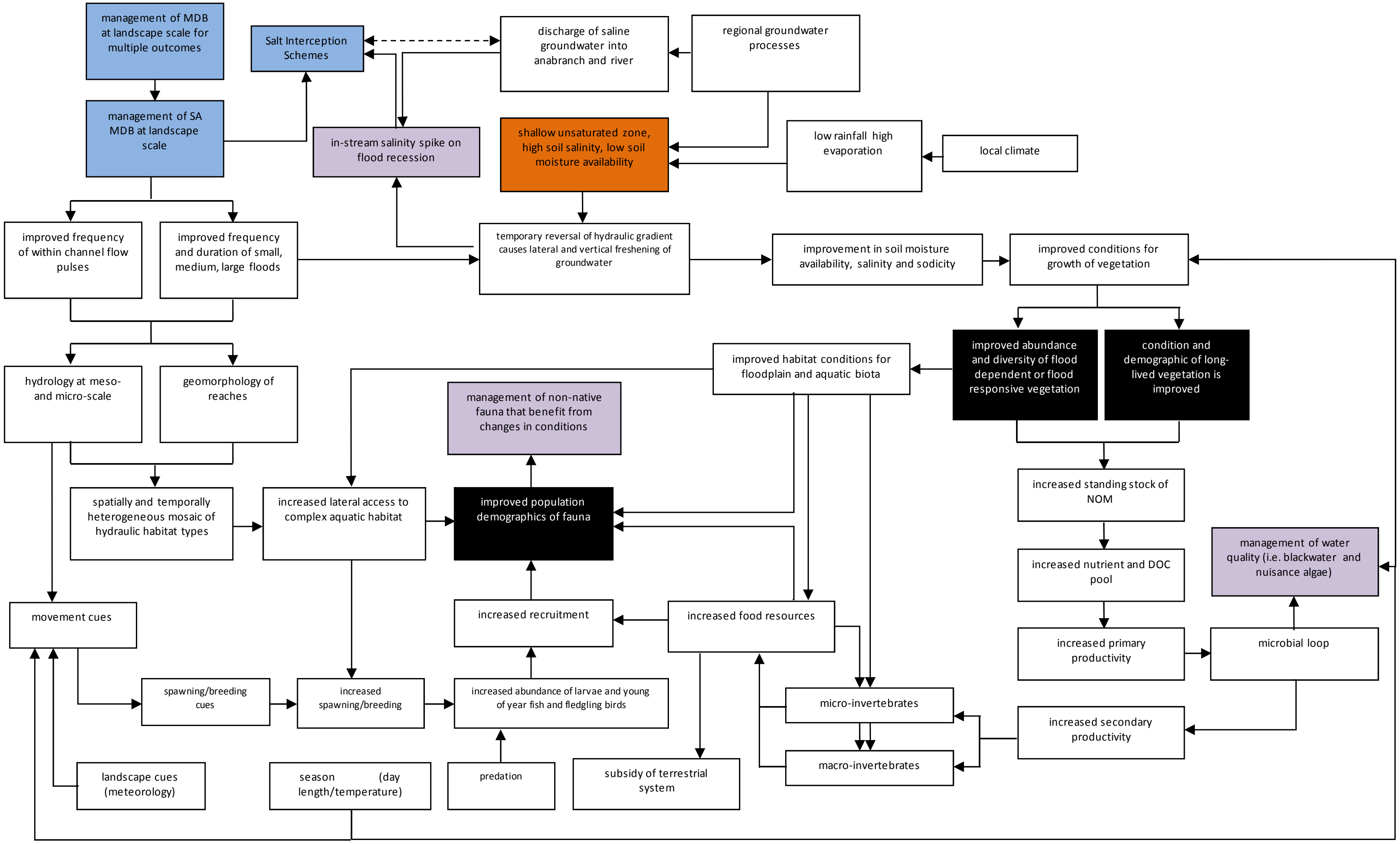

Figure 6.1 Simple conceptual model of the potential responses to the management levers of improved frequency of in–channel flow spikes and improved flooding frequency. Blue boxes represent direct management levers. Black boxes represent current focus of most monitoring programs within the MDB. Purple boxes represent potential negative outcomes that need to be managed. Adapted from Wallace (2011).

It is recommended that preliminary targets are established that are reviewed on a regular but frequent basis in order to allow new knowledge to be incorporated via the adaptive management process.

Selection of appropriate indicators

Patterns of species richness, abundance and distribution are often utilised as surrogates for ecological processes (see for example the condition monitoring plans for the Icon Sites within the Murray Darling Basin Authority's The Living Murray Program). However, they provide little information on how the ecosystem is functioning (Harris 1994). Ecological processes and biotic patterns may or may not be directly linked (Bunn *et al.* 1999; Bunn 1995) and effects on biota are generally only observed towards the end point of environmental degradation (Norris and Thoms 1999). Furthermore, monitoring only the biota is only likely to detect change if monitoring is continuous and includes those biota which are changing. Where there is a lag between disturbance/management intervention and biotic response or the organisms are insensitive to the disturbance/management intervention, crucial changes may not be detected. Alternatively, if a change is detected, the reasons for the change are not likely to be evident (Norris *et al.* 2007) and management will not be able to respond appropriately. Hence, understanding ecosystem processes is likely to be critical for understanding outcomes at higher trophic levels.

It is important to acknowledge that where semi–quantitative sampling techniques are used, targets based on numerical abundance may confuse management as they may not be able to account for natural variations in abundance. Semi–quantitative techniques such as single–pass electrofishing (Kennard *et al.* 2001) are recognised as having high variability in catch efficiency dependent on prevailing conditions such as water depth, flow velocity, channel morphology and turbidity. Very high inter–annual variation in catch–per–unit–effort has been recorded within the Victorian component of the Chowilla Floodplain Icon Site using standardised electrofishing surveys undertaken during the period 2002–10 (Wallace 2010). It is important to recognise that due to natural environmental variations, population size (expressed as relative abundance) can vary so widely that population size is rarely a reliable indicator of human influence (Karr 1999). Given that it will be essential to identify if changes are driven by changes to the hydrology or other ecosystem drivers, it is critical that variability in relative abundance (e.g. catch per unit effort) due to limitations associated with methodology are not mistaken as changes in the population. Hence it is essential to test the sensitivity of indicators such as percentage of native species and exotic individuals, and assemblage composition of the relevant communities, to the disturbance regime (Kennard *et al.* 2001).

Indicators must be able to be measured, interpreted (understood) by researchers and acted upon by managers (Ladson *et al.* 1999). Reliable indicators for detecting the effects of EWA's need to behave in ways that are not responsive to other, unmanaged factors (Downes *et al.* 2002). Potential examples are:

- assemblages of flood dependent plant communities may be expected to be present in higher relative abundance and/or diversity in years with an EWA compared to years without
- relatively high abundance of young of year fish of flow–responsive species may be expected in years with an EWA compared to years without
- soil salinity may be expected to be lower in years with an EWA compared to years without
- spatial extent and magnitude of freshwater lenses may be expected to be larger in years with an EWA than without
- Elevated concentrations of bioavailable carbon and nutrients may be expected in seasons with an EWA than without

The above examples can be readily converted into testable hypotheses. Understanding the "gap" between the magnitude of benefit derived from an unmanaged flood and a managed application of environmental water will be critical to successively improving outcomes. Multivariate analysis and/or regression modelling using generalised linear models (GLM) can be utilised to determine the strength of relationships between indicators and the applied management regime(s). Indicators that respond in an informative (e.g. linear or curvilinear) manner should be retained; those that do not respond at all or respond in a random manner should be discarded (Smith *et al.* 2001; Jansen and Robertson 2001). Indicators may be categorised into three groups (Cairns and McCormick 1992):

- Early warning indicators
    - signify impending decline in condition
- Compliance indicators
    - reveal position within, or deviation from, acceptable limits
- Diagnostic indicators
    - explain the movement within, or deviation from, acceptable limits

The need to manage for resilience

The management approach needs to focus on delivering a hydrological regime that allows for the achievement of ecological objectives, not simply returning a historical, long–term average flood return interval. Moving away from generic recommendations of reinstating more natural flood return frequencies (e.g. 1 flood every 2–3 years) towards an understanding of the relationships that underpin the long–term viability of ecological communities is critical to (i) build resilience and (ii) minimise the need for managed inundations over the long–term. This will require an understanding of relationships between vegetation condition, soil moisture availability, and groundwater conditions (Johns *et al.*, 2009).

Reinstating resilience is regarded as the most effective and pragmatic pathway to manage ecosystems in order to withstand future droughts and provide ecosystem services (Scheffer and Carpenter 2003; Scheffer *et al.* 2001; Bond *et al.* 2008; Folke *et al.* 2004). Resilience was defined by Holling (1973) as “a measure of the persistence of systems and of their ability to absorb change and disturbance and still maintain the same relationships between populations or state variables....”. Resilience has multiple attributes, but four aspects are considered critical (Walker *et al.* 2004 as cited by (Folke *et al.* 2004)):

- *Latitude*; the maximum amount the system can be changed and still reorganize within the same state.
- *Resistance*; how large a disturbance is required to change the current state of the system.
- *Precariousness*; how close the system is to a threshold that, if breached, makes reorganization difficult.
- *Cross–scale relations*; how the three attributes above are influenced by the states and dynamics of the system, at scales above and below the scale of interest.

Unregulated river systems are likely to have a very large degree of latitude and resistance, displaying a dynamic regime (Holling, 1973) with two distinct extremes (Scheffer *et al.*, 2001; Scheffer and Carpenter, 2003) in which wetlands are always flooding or drying (Kingsford *et al.*, 2010). Rather than the dry and the wet phase being two distinct states with characteristic dominant biota, there is only a single state with two alternative phases interspersed by floods and droughts (Colloff and Baldwin, 2010); the system will progressively revert towards the preceding condition once the disturbance (flooding or drying) is removed. If a driver (i.e. permanent inundation or very long drying) exerts sufficient pressure to exceed the threshold for change, a catastrophic (rather than smooth) transition to an alternate state can occur (Scheffer *et al.*, 2001; Scheffer and Carpenter, 2003). This concept is presented in Figure 6.2. River management has skewed river channels towards the left of this model (anti–drought) and floodplains to the right (engineered drought). Regulated river systems are therefore likely to be in an extreme state of precariousness.

With regard to the dramatic decline in condition of river red gums throughout the Murray–Darling Basin, Colloff and Baldwin (2010) suggest that these communities "have not yet reverted to an alternate stable state" implying that the system is approaching a tipping point. Assessments of population demographics in regionally relevant areas outside of the Chowilla Floodplain demonstrate that there is insufficient recruitment to sustain the existing forest and woodland communities (George *et al.* 2005; Wallace 2009; Aldridge *et al.* 2012). It is expected that a similar condition will be recorded at Chowilla Floodplain.

Floodplain eucalypts (*E. camaldulensis* and *E. largiflorens*) retain the majority of their seed in the canopy. Trees in poor condition have been shown to produce less seed than those in good condition (George *et al.*, 2005; Jensen *et al.*, 2008). Hence, once there is widespread loss of mature trees there is no soil seed bank for regeneration. Given the time required for these trees to reach maturity and develop structural traits (i.e. hollows) that provide habitat for dependent fauna, once lost, costly restoration with extremely long lag phases will be required to reinstate these "ecosystem engineers" (Colloff and Baldwin, 2010). Loss of forest/woodlands leads to increased likelihood of establishment of grassland areas (Scheffer *et al.*, 2001); conversely, in grassland areas, droughts reduce grass cover decreasing the likelihood of fires which are a key control on the establishment of perennial shrubs (Folke *et al.*, 2004). This situation where simply restoring the original environmental conditions (i.e. natural flow regime) is not likely to be sufficient to induce a switch back to the pre–existing condition and that conditions need to be established that create a second shift, back to the “desired” condition is known as hysteresis (Scheffer *et al.*, 2001).

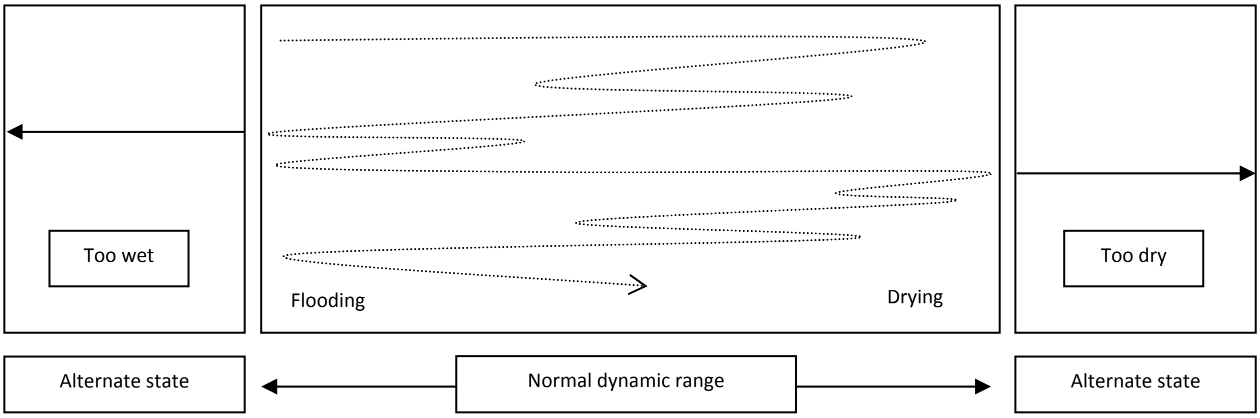

Figure 6.2 Conceptual model of the dynamic regime in which floodplains are always drying or flooding and the potential for excessive pressure to cause a transition to an alternate state. Source: Wallace *et al.* (2011).

7. Knowledge gaps

The purpose of this project was to scope out potential follow–on project(s) by providing an inventory of knowledge and knowledge gaps related to flows within the MDB and how they are affected by climate change, the Basin Plan, environmental flow management, the constraints and opportunities and relationship to SA departmental priorities for environmental and indigenous flow management in South Australia. This section provides an overall summary of knowledge gaps identified in this report and in recent Murray–Darling Basin Authority commissioned literature reviews on environmental watering for water birds (Brandis *et al.* 2009), trees (Johns *et al*. 2009), understorey and aquatic vegetation (Capon *et al.* 2009) and food webs (Brookes *et al.* 2009a).

7.1. Vegetation

The MDBA literature review on trees (Johns *et al*. 2009) considered the water requirements for 1) riparian and floodplain tree growth and health (including seed production), 2) propagule dispersal, 3) seed germination, and 4) seedling establishment. The high to medium priority knowledge gap hypotheses identified by Johns *et al*. (2009) were related to flooding regime characteristics (flood seasonality, longitudinal connectivity, rates of water level fall, flood duration, soil moisture, groundwater and flood depth) and tree health and reproduction (seedling growth and health, propagule dispersal during floods, dispersal of seeds, germination, reproduction (flowering to seed drop), adult health and growth and seedling growth and health). The medium to low priority knowledge gap hypotheses identified by Johns *et al*. (2009) were related to flooding regime characteristics (flood frequency, flood duration, water level rates of rise and fall, flood extent, flood seasonality and flow velocity) and tree health and reproduction (sapwood growth, seed settlement, seed dispersal, reproduction, germination and seed fall, seed dispersal and seedling growth and health) and physical conditions (soil moisture persistence and groundwater).

The MDBA review of understorey and aquatic vegetation (Capon *et al.* 2009) recommended that weed risk assessments were coordinated across all the icon sites to ensure that management of a particular weed or pest species at one site is not undermined by lack of control elsewhere and subsequent reintroduction. This review (Capon *et al.* 2009) identified knowledge gaps related to major life-history processes for understorey and aquatic vegetation.

- Occurrence of soil seed bank, vegetative propagule bank, vegetative propagule dispersal
- Effects of flow on plant growth, senescence, mortality, flowering, seed-set, seed dispersal, soil seed banks, seed germination, allocation to asexual reproduction, vegetative propagule dispersal, vegetative propagule banks, plant regeneration from vegetative propagules and establishment (e.g. from local mortality).

This review (Capon *et al.* 2009) also identified knowledge gaps related to vegetation dynamics for understorey and aquatic vegetation:

- Character & condition of seed banks, vegetative propagule banks;
- Effects of flow on seed bank character and condition, contribution of seed banks to vegetation regeneration, vegetative propagule bank character and condition, contribution of vegetative propagule banks to vegetation regeneration, propagule dispersal and plant-plant interactions; and
- Spatial and temporal patterns in propagule dispersal and the role of plant-plant interactions.

Priorities for research included developing i) a transferable plant functional group classification for use across all icon sites, ii) an understanding of the inherent temporal and spatial variability of key floodplain and aquatic vegetation communities and how this relates to hydrology, iii) conceptual models that make predictions concerning selected indicators with respect to current hydrological conditions that take into account this inherent variability and iv) preliminary limits of acceptable change in key indicators. They also recommended that experiments be used to determine the character and condition of propagule banks and their contribution to vegetation dynamics amongst key floodplain and aquatic vegetation communities as well as hydrological influences on these.

Other research questions regarding aquatic and floodplain understorey vegetation arising from this report that are specific to the lower River Murray are:

- The role of floodplain understorey and aquatic vegetation in riverine and terrestrial trophodynamics.
- Relationship between fish and physical habitat in the Lower Lakes.
- Role of the floodplain understorey and aquatic plants as habitat for small mammals, reptiles, frogs and birds.
- Water quality preferences, EWRs and salinity thresholds for *Ruppia* spp. (in particular *R. megacarpa*, which is now absent) in the Coorong and Murray Estuary.

7.2. Birds and mammals

To date, research effort has been made to understand the responses of well–recognised, iconic water–dependent fauna and floodplain vegetation to flow regimes – yielding a general focus on fish, frogs, waterbirds, and *E. camaldulensis*, particularly in the River Murray system (MDBC 2008). Terrestrial fauna responses have received less attention, despite their significant contribution to biodiversity and ecological function in floodplain river ecosystems, and the likely benefits they receive from flood–driven productivity. Overall, the 'terrestrial' or dry phase of the floodplain has been relatively neglected in policy and management plans, as well as in scientific research. We suggest that achieving balanced understanding of these systems will enable more effective management for ecosystem sustainability (on and off the floodplain) in the long term.

Floodplain productivity is thought to be crucial for maintaining diversity and abundance of endothermic groups such as mammals and birds (Bunn *et al.* 2006), but this has rarely been quantitatively assessed (Robinson *et al.* 2002; Baxter *et al.* 2005; Ballinger and Lake 2006). Most studies of consumption of aquatic prey by terrestrial predators have been conducted in highland streams (e.g. Nakano and Murakami 2001; Iwata *et al.* 2003), with very few documenting food web interactions in lowland or dryland floodplain systems (Ballinger and Lake 2006). Further studies in lowland rivers such as the Murray exploring food web links between flooding, vegetation response, and fauna would be beneficial for both scientific understanding and management.

Research priorities have been identified in the MDBA literature review of environmental watering for water birds (Brandis *et al.* 2009) and for woodland bird communities that use the river valley (McGinness *et al.* 2010). Water bird knowledge gaps include: structure of landscapes that maintain breeding populations and responses to changes in floodplain water regimes at broad landscape scales; wetland scale habitat use (i.e. changes in the availability of different habitat types that affect waterbird abundance and diversity); hydrological cues for foraging and breeding, waterbird movement and habitat use, importance of shallow (<50cm) wetland areas (that provide habitat for waterbirds and breeding floodplain fish) compared to the main river channel (that does not support the diversity of water regimes or extent of habitats and food needed to promote recruitment in fish and bird populations). Knowledge gaps related to woodland bird communities include: quantification of the links between flooding, food availability, and woodland bird breeding effort, breeding success, abundance, diversity, and community composition; quantification of any lags in woodland bird response to flooding or changes in flooding; tracking studies using marked birds to explore movements of bird species in response to flood events and increased primary productivity; and metapopulation studies exploring the influence / subsidy effect of floodplain productivity on bird populations at landscape scales, including dispersal and food web research.

7.3. Fish

A suite of knowledge gaps have been identified relating to fish ecology and the response of fish to flow variability in the lower River Murray, through both review of literature and based upon expert opinion. Knowledge gaps relate to three broad categories; 1) general biology (e.g. habitat associations, metapopulation dynamics), 2) the influence of flow on critical life-history processes (e.g. recruitment) and 3) the influence of river operations and engineering interventions on fishes.

Whilst there is significant knowledge of the general biology of most species of fish in the lower River Murray much remains unknown. Such knowledge is fundamental to managing fish populations and in planning and managing environmental flows. Indeed significant knowledge gaps exist regarding the habitat preferences of certain species and metapopulation dynamics in the lower River Murray. For instance, the status of a species may be influenced by movement within the lower River Murray and between the lower River Murray and other areas (e.g. the lower Darling). Knowledge of such spatial interaction is limited. Specific knowledge gaps related to the general biology of fishes of the lower River Murray include:

- Hydraulic and physical habitat preference of fish species at different life history stages;
- Lateral movement and off–channel habitat use by native and invasive species under different flow conditions;
- The importance of inundated floodplains for fish spawning and recruitment.;
- Habitat associations of fishes of the Lower Lakes and the influence of flow variability on these habitats, populations and associations;
- Movement and metapopulation dynamics of Murray cod and golden perch in the lower River Murray and lower Darling River;
- Movement and metapopulation dynamics of fish species across the Lower Lakes and lower River Murray; and
- Movement and metapopulation dynamics of key estuarine and diadromous fish species across the Lower Lakes, Coorong and the sea. i.e. What is the importance of the Coorong to broader fish population dynamics?

Flow is a key driver of critical life-history processes in fish, both directly and indirectly. An understanding of how flow influences life-history processes, such as spawning and recruitment, directly informs environmental flow delivery. Key knowledge gaps related to the influence of flow on critical life-history processes include:

- Environmental factors and mechanisms that influence spawning and recruitment success of flow critical species (i.e. golden perch, silver perch, Murray cod, etc) i.e. How do hydraulic conditions influence the recruitment of flow critical species?; and
- Environmental factors and mechanisms that influence spawning and recruitment success of estuarine dependent/associated species (e.g. black bream, greenback flounder, etc) and freshwater flow requirements for the Murray Estuary and Coorong.

River regulation has had a profound impact on the biota of the lower River Murray (Walker and Thoms 1993). Nonetheless, the use of the current weir network to vary water levels and large-scale floodplain regulators (e.g. the Chowilla Regulator) to enhance the frequency of floodplain inundation is at the forefront of environmental management. This is despite limited knowledge of the influence of these operations on fishes of the lower River Murray. Key knowledge gaps related to river operations and engineering interventions and their interplay with flow variability include:

- Cumulative effects of floodplain regulators on native and alien fish in the lower River Murray;
- An understanding of the links between hydrological and hydraulic conditions in the main channel of the lower River Murray; and
- Mechanisms for restoring hydraulic complexity in the lower River Murray.

7.4. Macro– and micro–invertebrates

Macro– and micro–invertebrates are a key link in energy transfer from carbon and nutrient dynamics → primary productivity → to higher level organisms. The role of micro biota (protists, rotifers, micro crustaceans) emerging from temporary wetlands and floodplains on the recruitment success of native fish species is a knowledge gap. Understanding the links between hydrological cues that trigger fish spawning responses and hydro-ecological processes that result in improved survival of larval and young-of-year stages is required in order to quantify the ability of management actions at the reach or floodplain scale to influence foodwebs at the regional scale. Within this, it will be important to quantify the influence of season and water quality (temperature, salinity, nutrients, dissolved oxygen) on the development of invertebrate communities during natural periods of high flows/floods and managed environmental water. Research priorities include the effect of:

- low habitat fragmentation due to permanent connectivity between river channel, anabranch and wetlands on the diversity of food resources available to higher level consumers; and
- high habitat fragmentation due to the reduced frequency of connection of the floodplain and the river on the status of invertebrate seed banks and hence diversity of food resources available to higher level consumers.

7.5. Carbon, nutrients, algal blooms and biofilms

The MDBA literature review on foodwebs (Brookes et al., 2009a) reviewed the state of knowledge and identified a number of knowledge gaps and priority areas for research. Those priorities, and others identified as part of this review, have been consolidated into the following outline of key areas where management would benefit from integrated research. What is clear, is that in order to achieve ecosystem restoration in the lower River Murray, managers will need to achieve a return to a state of efficient resource processing and energy transfer (Brookes *et al.*, 2009a; Brookes et al., 2005). This involves a number of processes that can be divided into groups, but in reality are intertwined in complex series of cascades and feedback loops. The importance of many of these processes and the need to manage hydrology to achieve these objectives can be difficult to convey to the general public. However, they are a key link in the transfer of resources (carbon and energy) from base sources to higher level animals that the public can more readily relate to.

A key priority for improved understanding and management of carbon and nutrient dynamics is the development of a conceptual model that is specific to Australian lowland rivers. This will improve (i) capacity for framing research hypotheses and linking hydrology-ecology relationships, and (ii) conveying the need to manage for these processes and communicating the outcomes from environmental watering activities to the wider community.

Areas requiring a combination of monitoring and research across a range of flows include determining the fate of allochthonous material on riverine foodwebs. A priority area for research is to determine if the large store of carbon that is released from inundated material into the water column is (i) cycled into higher trophic levels, or (ii) respired through the microbial loop. Within this, it will be important to understand the role of flows of different magnitudes, but also responses to other hydrology metrics altered by river regulation and environmental water delivery such as season, duration, return frequency and antecedent condition. It will be important to determine if there is a duration threshold (e.g. >30 days) and/or seasonal threshold (e.g. August/September) that significantly alters the magnitude of ecological benefit gained from environmental water. A key management question that would be answered by such a hypothesis is "Are environmental flows that occur in winter capable of producing ecological benefit to foodwebs?" This is a critical question, as delivering flows in winter is often proposed as a means of reducing water quality risks and the likelihood of carp spawning during managed inundations using constructed infrastructure.

Studies on the benefits and impacts of blackwater events are also required. Although blackwater events are generally regarded by environmental managers and the community as a negative outcome and something to be avoided, they are a natural occurrence, and the spike in availability of carbon and nutrients resulting from floods is likely to be a crucial component of ecosystem function. Desktop level priorities include further refinement of predictive models that allow managers to adaptively manage flows to avoid generating such events during use of environmental water. A logical extension of this is to create monitoring capacity in the form of establishing dissolved oxygen loggers at strategic points in the river to allow real time tracking of conditions and thereby improving capacity to manage water quality during environmental flows. Research priorities include:

- identifying the positive impacts of blackwater on aquatic food webs; and
- increasing understanding of the sub-lethal physiological and behavioural responses of aquatic biota to hypoxic blackwater.

There is often an assumption that meeting the hydrology metrics for EWRs is all that is required to achieve ecological outcomes. For flows that exceed the influence of the ten weirs in the lower River Murray, meeting discharge metrics is likely to improve the condition of targeted attributes (i.e. vegetation) on the floodplain. However, at lower flows (i.e. <40,000 ML/day) the serial weirs and their routine mode of operation (stable water levels) mean that the behaviour of parameters such as depth, lateral connectivity, velocity and turbulence have been largely disconnected from the way they behaved in response to the same discharge under natural conditions. Increasing our understanding of these complex hydraulic processes and ecological outcomes associated with delivery of environmental water and management of in-stream structures (i.e. weir pool lowering and raising) is a key priority. The role of areas of low flow velocity as sites of sedimentation for resources (carbon, nutrients, plant propagules, drifting larvae of invertebrates and fish) and the subsequent loss (microbial decomposition) of those resources is a knowledge gap. Within this, it is highly likely that the source of water used will have a substantial influence on the ecological benefit derived from environmental water. The range of different sources of water includes:

- water delivered via in-channel flows from upstream storages;
- water derived from upstream flooding (managed and natural flooding);
- regional flooding; and
- localised managed flooding at individual sites using constructed infrastructure.

The impact of water source on ecological outcomes needs to be taken into account not only to understand ecological outcomes from specific management actions, but also in the planning phase for delivery of environmental water. Management actions at upstream sites may reduce water quality (i.e. dissolved oxygen, salinity) such that planned actions at downstream sites need to be reduced in magnitude to avoid or manage serial impacts.

There is an existing body of work that identifies the processes involved in algal bloom development, substantial growth of the population and a sustained bloom in the lower River Murray. River mixing criterion and existing data sets indicate that periods of persistent thermal stratification are rare in the lower River Murray. However, key knowledge gaps associated with increasing availability of environmental water include the effect of return flows from managed inundations using constructed infrastructure into a low flowing river. This includes the co-ordinated management (e.g. drawdown) of a large number of wetlands during a weir pool drawdown. Furthermore, the impact of multi-reach weir pool raisings in spring-early summer that could result in flow velocity falling below the mixing criterion and allowing the establishment of persistent stratification needs to be determined.

Biofilm composition and food quality has been substantially altered by river regulation and stabilisation of flows. The impact of (i) historical de-snagging practices and (ii) water column stabilisation resulting from flow management on reducing the area of substrate available to support biofilm communities is a knowledge gap that may inform the need for re-snagging projects to improve the availability of complex submerged habitat for multiple benefits. The relative role of carbon and nutrients from biofilms as a food resource compared to other autochthonous (phytoplankton, submerged and emergent macrophytes) and allochthonous (e.g. eucalypt leaf litter and floodplain plant material) sources are knowledge gaps. Key research priorities include determining the ability of the available management actions to improve the successional status and food quality (e.g. C:N ratio) of biofilms. Management actions that could be tested to evaluate their capacity to improve biofilm quality include:

- intermittent periods of desiccation (weir pool lowering);
- increasing scour via increasing velocity (weir pool lowering, high flows); and
- altering the light environment that biofilms experience by (i) increasing the depth of submergence (weir pool raising and high flows) or (ii) increasing turbidity by using different water sources (upper Murray, Lake Victoria, Darling River).

7.6. Indigenous flows

There exists a growing body of Australian literature addressing knowledge gaps in non-Indigenous understandings of Indigenous conceptualisations of water and the impediments to Indigenous involvement in water research, planning and policy (for example see: Hemming *et al.* 2002; Morgan *et al.* 2004; Hattam *et al.* 2007; Jackson 2007, 2012). This scoping study provides an overview and assessment of these gaps as they relate to Indigenous flows and environmental flows in the Murray-Darling Basin.

Highlighted in this report are Indigenous innovations in engagement strategies being developed in South Australia. These innovations are focussing on new partnerships between Indigenous nations, government and researchers that address the incorporation of Indigenous interests into water research, policy development and planning. A significant knowledge gap in this arena relates to Federal government and broader researcher knowledge of Indigenous-driven, engagement strategies aimed at supporting overall Indigenous community wellbeing (see Hemming *et al.* 2011). This holistic approach requires ‘water’ to be understood as fundamentally connected to Indigenous social, cultural, spiritual and economic wellbeing (see Birckhead *et al.* 2011). From an Indigenous standpoint, Indigenous nations across the Murray-Darling Basin possess an uneven knowledge of government water management regimes and often limited capacity to participate, or take a leadership role in the management of their lands and waters (see Jackson 2012).

7.7. Predictive ecology

Whilst some modelling tools have found to be useful in decision-making, most of these are static index-based models, that only poorly represent dynamics, and are not posed in a systems framework. Despite multi-million dollar investments in data collection, translating the outcomes from data collection to improved ecological response modelling remains a persistent knowledge gap. Major areas of improvement for models are in:

- advancing environmental water requirements beyond simplistic paradigms. Water requirements are often assumed to based on linear or threshold based relationships and simple causal relationships;
- predicting outcomes dynamically. Models currently lack representation of spatial and temporal change. Ecosystem-based mapping information tends to be ad hoc and infrequently collected. Hydrologic models, including inundation models are often too coarse and too simple; and
- representation of connected landscapes, including river-floodplain-terrestrial ecotopes. Models poorly represent connectivity between vegetation communities, and the role of this within a systems framework.

Models for fish species and populations tend to be based on simplistic representations, where habitat is used as a proxy for populations (Shenton *et al.* 2012). Contemporary fish modelling approaches for the River Murray are still based on the Murray Flow Assessment Tool (Young *et al.* 2003). There has been limited testing of this tool, and it has been demonstrated to be insensitive to alternative flow scenarios, poorly representative of extreme flow events and model outcomes do not represent sequencing of events in any depth. Demographic based models are being advocated as a replacement of habitat models as they tend to more grounded in ecological theory. However, there is resistance to this approach based on data availability. As stated in Shenton *et al.* (2012) “whilst data requirements of demographic models will undoubtedly expose gaps in existing knowledge, but, in so doing, will strengthen future efforts to link changes in river flows with their ecological consequences”. In aiming to simply refine MFAT tools, we miss the opportunity to advance methods and continue to poorly represent ecological outcomes, as demonstrated in Lester *et al.* (2011).

In the case of the lower River Murray, there is a need for modelling tools that take into account the effect of salinity on floodplain vegetation responses to flooding that can be applied at a landscape scale. This then leads to a need to better understand the effect of spatial shifts in plant communities towards more drought tolerant species occupying the low–lying, ‘active’ floodplain under reduced future flooding frequencies. It is recommended that the following tasks be undertaken to develop this modelling capability.
Revise the most recent (2002–03) landscape scale vegetation survey of the lower River Murray floodplain to accurately assess current vegetation condition, taking account of recent developments in vegetation survey techniques. This should be combined with remote sensing and flood inundation modelling to identify trajectories of change in response to drought and salinity that can be used to model future floodplain vegetation communities.

- Use the vegetation survey data to develop vegetation response curves for different salinity levels (e.g. low, medium and high), soil types (e.g. sand, sandy clay, clay) and tree species using a water balance approach based on existing calibrated WAVES and WINDS modelling, vegetation health surveys and existing airborne geophysics derived groundwater salinity mapping.
- Determine the effect of current poor vegetation health on the ability of the individual species to recover and whether additional rehabilitation is required. This should include vegetation species recruitment and community succession related to reduced plant water availability caused by increased salinity and / or reduced flooding.

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Appendix 1. Workshop report

A stakeholder workshop was held on Tuesday 3rd April, at the SARDI centre at West beach, Adelaide. A list of attendees is given at the end of this appendix.

Aims:

- To identify potential priority project(s) relating the River Murray for Goyder funding
- To prioritise, with reasons, projects that are most important and urgent for the support of policy development and management of the River Murray
- To develop a follow–up process for testing and refining project plans for the priority project(s).

Program:

Timing	*Session*	*Description*	*Action*
9:00 – 9:05	Introduction	Introduce the aims and process for the workshop. Describe why this group of individuals were invited	Paul
9:05 – 10:25	Description of potential projects	Describe the broader Scoping Study, who is doing it, timeframes, objectives Describe the priority projects identified in the Goyder R&D Plan: 30 minutes for audience response. The Roadmap is largely settled, but are there any obvious gaps, or have some of these issues been dealt with through other processes?	Mac Paul + presenters
10:25 – 10:40	Morning Tea		
10:40 – 11:20	Prioritisation	In groups of 4–5, identify the three highest priority projects. For these three, fill in the Proforma, and be prepared to present back. Prioritisation should be based on the potential of the research to create an impact: • There is an existing knowledge gap • improve existing policy, • improve existing management • doesn't duplicate other projects/processes • feasible to deliver impact with budget and time available	Paul
11:20 – 12:10	Report Back	Plenary session to collate all scores and select final priority project (if obvious), or selection of priority projects with reasons why each is important. It has been agreed that the scoping project should recommend one follow–on project, but a report back will also indicate which were the next ranked projects in case the Goyder or RAC wants another choice. It has been agreed that if two similar projects were equally top–ranked (eg b. and c. – hydrology scenarios and climate change impacts in the list above), we would discuss with the workshop participants that they might be combined.	Paul

Timing	*Session*	*Description*	*Action*
12:10 – 12:25	Discussion	Discussion of overall direction of top ranked project(s). Does this pass the common sense test? Are we all in agreement?	Paul
12:25 – 12:30	Next Steps	Discussion of follow–up process 1) How will the information from this workshop be fed back? 2) How will this group be informed of the final decision of the RAC and Board? 3) Do they want another workshop, for example? For what purpose?	Paul
12:30 – 1:00	Lunch		

Discussion of potential projects:
In this session, the ten potential projects suggested by the Goyder Institute itself when it commissioned the project were described by the team members. The workshop also gave feedback for each proposed project, which in the case of the first and the last removed them from further consideration. At the end of the session, the audience were invited to comment on whether there were any missing projects or other obvious gaps.

A. **Water resource accounting tool**
   An excel tool is already used to account for water from the Commonwealth so this topic has been removed from further discussion

B. **Hydrological Scenario Development**
   Take a model which already exists (eg the simplified MDB model being developed by Dushmanta Dutta) and develop a series of scenarios and provide outcomes. For example – what does climate change mean to water delivery or what is the impact in SA if a change in water delivery upstream occurs?

C. **Impacts of climate change on River Murray Flows**
   Under a median climate change scenario, it is projected that there will be around 2500GL less in the system but 2030. What does this mean to SA? This could be modelled using tools such as the CSIRO simplified MDB model.

D. **Effective implementation of the MDB Plan – socioeconomic tradeoff–benefits**
   SA socioeconomic and cultural factors, Impacts upstream of available water upstream and the Basin–wide socio–economic impacts. Need to defend how water is being managed.

E. **River Murray and Floodplains, including salt and in the landscape (referred to as "Floodplain Processes")**
   Floodplain processes and connectivity of carbon, nutrients and biota. Role of floodplains. What floodplain do we want? What do they do? Indigenous aspects. Groundwater–surface water interactions. New methods to monitor ET, new methods to provide broad scale monitoring

F. **Ecological restoration processes**

Management plans don't identify ecological targets, so less focus on ecological objectives. Little process information on ecological flows, no focus on processes at the landscape scale (system, reach). Currently cater to patchwork habitats – risk of applying same rules t different areas will result in similar asset structure (whether in stream or out). Where does water come from – water from Darling VERY different to water from Dartmouth Dam in relation to turbidity, nutrients, clarity, temperature etc – so may not see outcome required due to water conditions. Delivery of water – 90% of ewater is held in wetlands instead of returning the floodplain benefit to the channel. Concepts of lateral and vertical connectivity, 1D, 2D and 3D movement from floodplain back to the river (or sky in terms of birds). Scale – hydrological and ecologic processes occur upstream and not just in SA – need to consider these also.

G. Engineered structures

Engineering structures should be part of toolkit– but how do we use these to maximise ecological outcomes? Engineering structures also increase suitability for invasive species (Carp).

H. Ecological/hydrological response model

Translation of hydrologic scenarios to ecological outcomes – Murray Ecology Project looks at the key ecologic component functions – next step is a response model. Use model to inform on how best to use a certain amount of water for a wetland or a floodplain or to create a flood pulse. Flow management tool within an adaptive management framework to inform policies. Suggest a combination of e,f, and g is required. We have a poor understanding on floodplain processes and functions but a good selection and understanding of data/ecological processes. What's the processes interacting between floodplain and channel? Landscape scale and consider spatial and temporal changes

I. System understanding of SA River Murray: an inventory

Find knowledge gaps. Understanding scale. Lots of data and lots of knowledge but how to we take all of that to create a broader understanding. If make changes in NSW and Vic what's the impact in SA? This is not so much a project but a philosophy so it really constitutes a body of work within a larger project. Holistic systems understanding already underway for the Snowy River (Grant)

J. Riverbank collapse

Very specific project – interaction between riverbank and hydrologic flows. Low river levels lead to riverbank collapse which has a significant impact in the vicinity of bridges and other structures. Project already expressed interest from Adelaide University and University of Sydney

Any missing projects?

The discussion brought out the following points:

A. Links between social outcomes and ecological outcomes – what are the co–benefits rather than the trade–offs? Does water go to the irrigator or the environment? How to relate environmental benefits to the irrigators to placate them?

B. Recognise that any project has an indigenous perspective – what process can Goyder put in place to ensure indigenous interaction?

C. Lack of water requirements for the River Murray Channel

D. Risks perceived with ewater management – blackwater events if water timing is wrong, what's the risk of downsizing the river system under the current Basin Plan since out floodplain areas will need to be sacrificed – impact of flow of energy, carbon, nutrients etc – understanding floodplain functions.

E. A second project to look at models which can feed in the socio–economics

Outcome:
Following the presentations and discussion of potential projects, the workshop broke up into tables, and discussed priorities amongst the projects. Table by table rankings were then reported back to the whole groups. The following table resulted:

A	
B	2
C	
D	2
E	3+2 ***
F	**
G	***
H	3****
G+H	3+2
B, E +H	3
G, E+H	3
G, E, H+F	3

*Relates to how many times the project appeared in conjunction with another project

The result of the scoring was not clear cut, but showed a clear leaning towards an ecologically oriented project, probably linked to river flows.

The scoring formed the basis for the ensuing discussion, which emphasized landscape connectivity, and the need to link ecology to flows. A clear consensus emerged for a project around floodplain processes and how floodplains interact with the river and terrestrial environment. It must include landscape scale and connectivity.

Additional points noted during the discussion were:

- Need to build a better understanding of processes to develop policy on EWR's – emphasis on collection of new information
- Want a product we can use as a management product
- Aim for a tool but not a necessary outcome for the Priority project – make it a "Project 2" follow–on
- Management objectives are the key we are trying to achieve
- Identify good empirical data areas and areas where data lacks – directed and targeted research
- We need to understand the end point of the project. As long as science is driven toward the objective if a tool.. .. we have data and understandings that can easily be translated to management objects, but a tool eventually would be good.
- Identify PhD topics which can feedback into the Priority project – describe areas where a PhD can focus
- Embed policy people in the scoping of the Priority project with constant involvement over the 2–3 year term
- Project team
    - Missing bird expertise
    - Operational side of managing rivers – Kane Aldridge?

- DENR?

Attendees:
Stakeholders
Adam Chambers, PIRSA
Chrissie Bloss, DfW
Daniel Rogers, DENR
Grant Rigney, Chair, Murray Lower Darling River Indigenous Nations
Isobel Campbell, Chair, Mannum Aboriginal Cultural Heritage Association
Jonathan McPhail, PIRSA
Kane Aldridge, DfW
Kelly Marsland, SA MDB NRM
Kumar Savadamuthu, DfW
Lachlan Sutherland, Aboriginal Project Officer, DENR
Luke Trevorrow, Ngarrindjeri Regional Authority
Michelle Bald, DfW
Neil Power, DfW
Nicholas Souter, DfW
Nigel Rutherford, SAWater
Nita Campbell, Mannum Aboriginal Cultural Heritage Association
Peter Goonan, EPA
Rebecca Turner, DENR
Stephen Campbell, DPC
Tim Hartman, Chair of Ngarrindjeri Ruwe Contracting and NRA Yarlwuar–Ruwe Program
Tom Trevorrow, Chair, Ngarrindjeri Regional Authority
Tony Minns, Goyder Institute
Tracey Steggles, DfW
Tumi Bjornsson, DfW
Vic Hughes, MDBA
Wei Yan, DfW

Project Team
Tanya Doody
Steve Hemming
Mac Kirby
Jason Nicol
Todd Wallace
Qifeng Ye
Brenton Zampatti

Facilitator
Paul Dalby

Appendix 2. EWRs for South Australia

Table A.1. Summary of Environmental Water Requirements for the South Australian reach of the River Murray. Source: adapted from Lamontagne (2012, Appendix A).

Source and #	Target	Environmental Water Requirement		
		Flow (ML d^{-1})	Duration (days)	Timing
SA–a1 (BBr1)	Successful recruitment of cohorts of black box at lower elevations	85,000	20	Spring or early summer
SA–a2 (BBr2)	Successful recruitment of cohorts of black box at higher elevations	>100,000	20	Spring or early summer
SA–b (BB1)	Maintain and improve the health of 80% of the black box woodlands	>100,000	20	Spring or summer
SA–c (BB2)	Maintain and improve the health of ~60% of the black box woodlands	100,000	20	Spring or summer
SA–d (BB3)	Maintain and improve the health of ~50% of the black box woodlands	85,000	30	Spring or summer
SA–e(RGr)	Successful recruitment of cohorts of river red gums	80,000	60	Aug – Oct
SA–f (RG)	Maintain and improve the health of 80% of the river red gum woodlands and forests (adult tree survival)	80,000 to 90,000	>30	Jun – Dec
SA–g (Lig1)	Maintain and improve the health of ~50% of the lignum shrubland	70,000	30	Spring or early summer
SA–g (Lig2)	Maintain and improve the health of 80% of the lignum shrubland	80,000	30	Spring or early summer
SA–h (Ligr)	Lignum shrubland recruitment – 66% of community maintained [2]	70,000	120	–
SA–I (Mos1)	Provide mosaic of habitats (i.e. larger proportions of various habitat types are inundated)	90,000	30	Spring or early summer
SA–j (Mos2)	Provide mosaic of habitats (i.e. larger proportions of various habitat types are inundated)	80,000	>30	Spring or early summer
SA–k (Mos3)	Provide mosaic of habitats (i.e. larger proportions of various habitat types are inundated)	70,000	60	Spring or early summer
SA–l (Mos4)	Provide mosaic of habitats (i.e. larger proportions of various habitat types are inundated)	60,000	60	Spring or early summer
SA–m (WB1)	Maintain lignum inundation for waterbird breeding events	70,000	60	Aug – Oct
SA–m (WB2)	Provide habitat (river red gum communities) for waterbird breeding events	70,000	60	Aug – Oct
SA–n (FP)	Stimulate spawning, provide access to the floodplain and provide nutrients and resources	80,000	>30	Jun – Dec
SA–o (TW1)	Inundation of (~80%) temporary wetlands for large scale bird and fish breeding events	80,000	>30	Jun – Dec
SA–p (TW2)	Maintain and improve majority of lower elevation (~20%) temporary wetlands in healthy condition; and Inundation of lower elevation temporary wetlands for small scale bird and fish breeding events, and microbial decay/export of organic matter	40,000	90	Aug – Jan
SA–q (FV)	Provide variability in flow regimes at lower flow levels	Pool to 40,000[3]	Variable	

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[2] An EWR for lignum recruitment was not provided in DWLBC (2010), however Goyder recommend the inclusion of a lignum recruitment target. This EWR has been developed from information provided in Ecological Associates (2010), cited in DWLBC (2010).

[3] While specific flow is defined, this EWR has been assessed as the percentage of years in which 40,000 ML day^{-1} is reached with 1 day minimum duration

Table A. 2. Summary of Environmental Water Requirements that have been developed for the South Australian reach of the River Murray. Source: adapted from Lamontagne (2012, Appendix A).

No.	Target	Environmental Water Requirement Flow ($ML\ d^{-1}$)	Duration (days)	Notes About Requirement Timing (season)	Min Duration (days)
MDBA 1	Freshes	20,000	60	–	Longest single continuous
MDBA 2	Maintain 80% of the current extent of wetlands in good condition	40,000	30	Jun – Dec	7
MDBA 3	Maintain 80% of the current extent of red gum forest in good condition	40,000	90	Jun – Dec	7
MDBA 4	Maintain 80% of the current extent of red gum forest in good condition	60,000	60	Jun – Dec	7
MDBA 5	Maintain 80% of the current extent of red gum forest in good condition, maintain 80% of the current extent of red gum woodland in good condition	80,000	30	Pref. winter/spring but timing not constrained	7
MDBA 6	Maintain 80% of the current extent of black box woodland in good condition	100,000	21	Pref. winter/spring but timing not constrained	1
MDBA 7	Maintain 80% of the current extent of black box woodland in good condition	125,000	7	Pref. winter/spring but timing not constrained	1

Appendix 3. Understory plant species list for the South Australian River Murray Corridor

(*denotes exotic species, **denotes proclaimed noxious in South Australia, *** denotes weed of national significance, # denotes listed as rare in South Australia, ## denotes listed as endangered in South Australia).

Species	Family	Functional Group	Native species associated with aquatic or floodplain ecosystems
*Abutilon theophrasti**	Malvaceae	Floodplain	
Acacia ligulata	Mimosaceae	Terrestrial dry	
Acacia myrtifolia	Mimosaceae	Terrestrial dry	
*Agapanthus praecox**	Amaryllidaceae	Terrestrial dry	
Alisma plantago–aquaticum	Alismataceae	Amphibious fluctuation responder–plastic	🟩
Alternanthera denticulata	Amaranthaceae	Floodplain	🟩
Alternanthera nodiflora	Amaranthaceae	Floodplain	🟩
Ammania multiflora	Lythraceae	Floodplain	🟩
Amyema melaleucae	Loranthaceae	Parasitc on floodplain trees	🟩
Amyema miquelii	Loranthaceae	Parasitc on floodplain trees	🟩
Amyema preissii	Loranthaceae	Parasitc on floodplain trees	🟩
*Anagallis arvensis**	Primulaceae	Terrestrial damp	
Angianthus tomentosus	Asteraceae	Terrestrial dry	
*Apium graveolens**	Apiaceae	Terrestrial damp	
Apium prostratum ssp. *prostratum*	Apiaceae	Floodplain	🟩
*Arctotheca calendula**	Asteraceae	Terrestrial dry	
*Asparagus asparagoides***	Liliaceae	Terrestrial dry	
*Asparagus officinalis**	Liliaceae	Terrestrial dry	
Asperula gemella	Rubiaceae	Floodplain	🟩
*Asphodelus fistulosus**	Asphodelaceae	Terrestrial dry	
*Aster subulatus**	Asteraceae	Amphibious fluctuation tolerator emergent	
Atriplex cinerea	Chenopodiaceae	Terrestrial dry	
Atriplex eardleyae	Chenopodiaceae	Terrestrial dry	
Atriplex holocarpa	Chenopodiaceae	Terrestrial dry	
Atriplex leptocarpa	Chenopodiaceae	Floodplain	🟩
Atriplex limbata	Chenopodiaceae	Terrestrial dry	
Atriplex lindleyi	Chenopodiaceae	Terrestrial dry	
Atriplex nummularia	Chenopodiaceae	Terrestrial dry	
Atriplex paludosa ssp. *cordata*	Chenopodiaceae	Terrestrial dry	
Atriplex paludosa ssp. *paludosa*	Chenopodiaceae	Terrestrial dry	
*Atriplex prostrata**	Chenopodiaceae	Floodplain	
Atriplex pseudocampanulata	Chenopodiaceae	Terrestrial dry	
Atriplex semibaccata	Chenopodiaceae	Terrestrial dry	
Atriplex stipitata	Chenopodiaceae	Terrestrial dry	
Atriplex suberecta	Chenopodiaceae	Floodplain	🟩
Atriplex velutinella	Chenopodiaceae	Terrestrial dry	
Atriplex vesicaria	Chenopodiaceae	Terrestrial dry	
Austrostipa puberula	Poaceae	Terrestrial dry	
Austrostipa stipoides	Poaceae	Terrestrial dry	
*Avena barbata**	Poaceae	Terrestrial dry	
*Avena fatua**	Poaceae	Terrestrial dry	
Azolla filiculoides	Azollaceae	Floating	🟩
Azolla pinnata	Azollaceae	Floating	🟩
Batrachium trichophyllum	Ranunculaceae	Floating	🟩
Baumea juncea	Cyperaceae	Amphibious fluctuation tolerator emergent	🟩
Berula erecta	Apiaceae	Amphibious fluctuation tolerator emergent	🟩
Boerhavia dominii	Nyctaginaceae	Terrestrial dry	
Bolboschoenus caldwellii	Cyperaceae	Emergent	🟩
Bolboschoenus medianus	Cyperaceae	Emergent	🟩
Brachyscome basaltica	Asteraceae	Floodplain	🟩
Brachyscome linearilobia	Asteraceae	Floodplain	🟩
Brachyscome dentata	Asteraceae	Floodplain	🟩

Species	Family	Functional Group	Native species associated with aquatic or floodplain ecosystems
*Brassica rapa**	Brassicaceae	Terrestrial dry	
*Brassica tournifortii**	Brassicaceae	Terrestrial dry	
*Briza minor**	Poaceae	Terrestrial dry	
Bromus arenarius	Poaceae	Terrestrial dry	
*Bromus catharticus**	Poaceae	Terrestrial dry	
*Bromus diandrus**	Poaceae	Terrestrial dry	
*Bromus hordeaceus**	Poaceae	Terrestrial dry	
*Bromus molliformis**	Poaceae	Terrestrial dry	
*Bromus rubens**	Poaceae	Terrestrial dry	
*Bromus tectorum**	Poaceae	Terrestrial dry	
*Bromus uniloides**	Poaceae	Terrestrial dry	
*Bupleurum semicompositum**	Apiaceae	Terrestrial dry	
Callistemon brachyandrus#	Myrtaceae	Amphibious fluctuation tolerator–woody	🟩
*Callitriche stagnalis**	Callitrichaceae	Amphibious fluctuation responder–plastic	
Calotis cuneifolia	Asteraceae	Floodplain	🟩
Calotis hispidula	Asteraceae	Floodplain	🟩
Calotis scapigera	Asteraceae	Floodplain	🟩
Calystegia sepium	Convolulaceae	Amphibious fluctuation tolerator–emergent	🟩
Carpobrotus rossii	Aizoaceae	Terrestrial dry	
*Carrichtera annua**	Brassicaceae	Terrestrial dry	
*Centaurea calictrapa**	Asteraceae	Terrestrial damp	
*Centaurium tenuiflorum**	Asteraceae	Terrestrial damp	
Centella asiatica	Apiaceae	Amphibious fluctuation responder–plastic	🟩
Centipeda cunninghamii	Asteraceae	Floodplain	🟩
Centipeda minima	Asteraceae	Floodplain	🟩
Ceratophyllum demersum#	Ceratophyllaceae	Submergent k–selected	🟩
Chamaesyce drummondii	Iridaceae	Floodplain	🟩
Chara fibrosa	Characeae	Submergent r–selected	🟩
*Chenopodium album**	Chenopodiaceae	Terrestrial damp	
Chenopodium curvispicatum	Chenopodiaceae	Terrestrial dry	
Chenopodium desertorum	Chenopodiaceae	Terrestrial dry	
*Chenopodium glaucum**	Chenopodiaceae	Terrestrial damp	
Chenopodium nitrariaceum	Chenopodiaceae	Terrestrial dry	
Chenopodium pumilio	Chenopodiaceae	Floodplain	🟩
*Chloris truncata**	Poaceae	Terrestrial dry	
*Chrysanthemoides monilifera****	Apiaceae	Terrestrial dry	
*Cirsium vulgare**	Apiaceae	Terrestrial damp	
*Citrullus lanatus**	Curcubitaceae	Terrestrial dry	
Clematis microphylla var. *microphylla*	Ranunculaceae	Terrestrial dry	
*Conyza bonariensis**	Asteraceae	Terrestrial damp	
Cotula australis	Asteraceae	Amphibious fluctuation responder–plastic	🟩
*Cotula bipinnata**	Asteraceae	Amphibious fluctuation responder–plastic	
Cotula coronopifolia	Asteraceae	Amphibious fluctuation responder–plastic	🟩
Cotula vulgaris var. *australasica*	Asteraceae	Amphibious fluctuation responder–plastic	🟩
Craspedia chrysantha	Asteraceae	Floodplain	🟩
Craspedia uniflora	Asteraceae	Floodplain	🟩
Crassula colorata var. *acuminata*	Crassulaceae	Amphibious fluctuation tolerator–low growing	🟩
Crassula helmsii	Crassulaceae	Amphibious fluctuation tolerator–low growing	🟩
Crassula sieberana##	Crassulaceae	Amphibious fluctuation tolerator–low growing	🟩
Cressa cretica	Brassicaceae	Floodplain	🟩
Crinum flaccidum	Amaryllidaceae	Terrestrial dry	
*Critesion marinum**	Poaceae	Terrestrial dry	
*Cucumis myriocarpus**	Curcubitaceae	Terrestrial dry	

Species	Family	Functional Group	Native species associated with aquatic or floodplain ecosystems
*Cuscuta campestris**	Convolulaceae	Parasitic	
*Cynara cardunculus**	Asteraceae	Terrestrial damp	
*Cynodon dactylon**	Poaceae	Terrestrial dry	
Cyperus difformis	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Cyperus exaltatus	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Cyperus gunnii ssp. *gunnii*	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Cyperus gymnocaulos	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Danthonia caespitosa	Poaceae	Terrestrial dry	
*Datura ferox**	Solanaceae	Terrestrial dry	
*Datura stramonium**	Solanaceae	Terrestrial dry	
Dianella brevicaulis	Liliaceae	Terrestrial dry	
Dianella revoluta	Liliaceae	Terrestrial dry	
Disphyma crassifolium ssp. *clavellatum*	Aizoaceae	Terrestrial dry	
Dissocarpus paradoxus	Chenopodiaceae	Terrestrial dry	
Distichlis distichophylla	Poaceae	Floodplain	🟩
*Dittrichia graveolens**	Asteraceae	Floodplain	
Dodonaea attenuata	Sapindaceae	Terrestrial dry	
Dodonaea viscosa	Sapindaceae	Terrestrial dry	
Duma florulenta	Polygonaceae	Amphibious fluctuation tolerator-woody	🟩
Duma horrida#	Polygonaceae	Amphibious fluctuation tolerator-woody	🟩
*Echinochloa crus–galli**	Poaceae	Terrestrial damp	
*Echium plantagineum***	Boraginaceae	Terrestrial dry	
Eclipta platyglossa	Asteraceae	Terrestrial dry	
*Ehrharta longiflora**	Poaceae	Terrestrial dry	
Einadia nutans ssp. *nutans*	Chenopodiaceae	Terrestrial dry	
Eleocharis acuta	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Eleocharis gracilis	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Eleocharis sphacelata	Cyperaceae	Emergent	🟩
*Elodea canadensis***	Hdyrocharitaceae	Submergent k–selected	
Enchylaena tomentosa var. *tomentosa*	Chenopodiaceae	Terrestrial dry	
*Enneapogon nigricans**	Poaceae	Terrestrial dry	
Epaltes australis	Asteraceae	Floodplain	🟩
Epaltes cunninghamii	Asteraceae	Floodplain	🟩
Epilobium pallidiflorum	Lythraceae	Floodplain	🟩
Eragrostis australasica	Poaceae	Floodplain	🟩
*Eragrostis curvula**	Poaceae	Terrestrial damp	
Eragrostis dielsii	Poaceae	Floodplain	🟩
Eragrostis lacunaria	Poaceae	Floodplain	🟩
Eremophila divaricata	Myoporaceae	Terrestrial dry	
Eremophila gibbifolia	Myoporaceae	Terrestrial dry	
Eremophila scoparia#	Myoporaceae	Terrestrial dry	
Eriochiton sclerolaenoides	Chenopodiaceae	Floodplain	🟩
*Erodium cicutarium**	Geraniaceae	Floodplain	
Euchiton involucratus	Asteraceae	Floodplain	🟩
Euphorbia drummondii	Euphorbiaceae	Floodplain	🟩
Euphorbia tannensis	Euphorbiaceae	Floodplain	🟩
*Euphorbia terracina**	Euphorbiaceae	Terrestrial dry	
Exocarpus aphylla	Santalaceae	Terrestrial dry	
*Foeniculum vulgare**	Apiaceae	Terrestrial damp	
Frankenia pauciflora var. *gunnii*	Frankeniaceae	Terrestrial dry	
Frankenia serpyllifolia	Frankeniaceae	Terrestrial dry	
Frankenia uncinata	Frankeniaceae	Terrestrial dry	
*Fumaria bastardii**	Fumariaceae	Terrestrial damp	
Fumaria capreolata ssp. *capreolata**	Fumariaceae	Terrestrial damp	
Gahnia filum	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩

Species	Family	Functional Group	Native species associated with aquatic or floodplain ecosystems
*Galenia secunda**	Aizoaceae	Terrestrial dry	
*Gazania linearis**	Asteraceae	Terrestrial dry	
Glinus lotoides	Aizoaceae	Floodplain	🟩
Glyceria australis	Poaceae	Amphibious fluctuation tolerator–emergent	🟩
Glycyrrhiza acanthocarpa	Fabaceae	Floodplain	🟩
Goodenia gracilis	Goodeniaceae	Floodplain	🟩
Haloragis aspera	Haloragaceae	Floodplain	🟩
*Helianthus annua**	Asteraceae	Terrestrial dry	
Helichrysum scorpoides	Asteraceae	Floodplain	🟩
*Heliotropium amplexicaule**	Boraginaceae	Floodplain	
*Heliotropium curassavicum**	Boraginaceae	Floodplain	
*Heliotropium europaeum**	Boraginaceae	Floodplain	
Hemichroa pentandra	Amaranthaceae	Floodplain	🟩
*Holcus lanatus**	Poaceae	Terrestrial dry	
Hydrilla verticillata	Hdyrocharitaceae	Submergent k–selected	🟩
Hydrocotyle verticillata	Apiaceae	Amphibious fluctuation responder–plastic	🟩
*Hypochaeris glabra**	Asteraceae	Terrestrial dry	
*Hypochaeris radicata**	Asteraceae	Terrestrial dry	
Iris sp.*	Iridaceae	Terrestrial dry	
Isachne globosa	Poaceae	Floodplain	🟩
Iseotopsis graminifolia	Asteraceae	Floodplain	🟩
Isolepis australiensis	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Isolepis cernua	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Isolepis fluitans	Cyperaceae	Submergent r–selected	🟩
Isolepis hookeriana	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Isolepis inundata	Cyperaceae	Submergent r–selected	🟩
Ficinia nodosa	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Isolepis platycarpa	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Isolepis producta	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Ixiolaena brevicompta	Asteraceae	Floodplain	🟩
*Juncus acutus**	Juncaceae	Amphibious fluctuation tolerator–emergent	
Juncus aridicola	Juncaceae	Amphibious fluctuation tolerator–emergent	🟩
*Juncus articulatus**	Juncaceae	Amphibious fluctuation tolerator–emergent	
Juncus caespiticius	Juncaceae	Amphibious fluctuation tolerator–emergent	🟩
Juncus kraussii	Juncaceae	Amphibious fluctuation tolerator–emergent	🟩
Juncus pallidus	Juncaceae	Amphibious fluctuation tolerator–emergent	🟩
Juncus pauciflorus	Juncaceae	Amphibious fluctuation tolerator–emergent	🟩
Juncus sarophorus	Juncaceae	Amphibious fluctuation tolerator–emergent	🟩
Juncus subsecundus	Juncaceae	Amphibious fluctuation tolerator–emergent	🟩
Lachnagrostis filiformis	Poaceae	Floodplain	🟩
*Lactuca saligna**	Asteraceae	Terrestrial dry	
*Lactuca serriola**	Asteraceae	Terrestrial dry	
*Lagunaria patersonii**	Malvaceae	Terrestrial dry	
*Lagurus ovatus**	Poaceae	Terrestrial dry	
*Lamarckia aurea**	Poaceae	Terrestrial dry	
Lamprothamnium macropogon	Characeae	Submergent k–selected	🟩
Lamprothamnium papulosum	Characeae	Submergent k–selected	🟩

Species	Family	Functional Group	Native species associated with aquatic or floodplain ecosystems
Lawrencia squamata	Malvaceae	Floodplain	🟩
Lemna disperma	Lemnaceae	Floating	🟩
Lemna minor	Lemnaceae	Floating	🟩
*Lepidium africanum**	Brassicaceae	Terrestrial dry	
*Lepidium bonariensis**	Brassicaceae	Terrestrial dry	
Lepilaena australis	Zannichelliaceae	Submergent r–selected	🟩
Lepilaena cylindrocarpa	Zannichelliaceae	Submergent r–selected	🟩
Leptospermum continentale	Myrtaceae	Terrestrial dry	
Lilaeopsis polyantha	Apiaceae	Amphibious fluctuation responder–plastic	🟩
*Limonium binervosum**	Limoniaceae	Terrestrial dry	
Limosella australis	Scrophulariaceae	Amphibious fluctuation responder–plastic	🟩
Lobelia alata	Campanulaceae	Floodplain	🟩
*Lolium rigidum**	Poaceae	Terrestrial dry	
Lomandra leucocephala ssp. *robusta*	Xanthorrhoeaceae	Terrestrial dry	
Ludwigia peploides ssp. *montevidensis*	Onagraceae	Amphibious fluctuation responder–plastic	🟩
*Lupinus cosentinii**	Fabaceae	Terrestrial dry	
*Lycium ferocissimum***	Solanaceae	Terrestrial dry	
Lycium australe	Solanaceae	Terrestrial dry	
Lycopus australis	Lamiaceae	Amphibious fluctuation tolerator–emergent	🟩
Lysiana exocarpi ssp. *exocarpi*	Loranthaceae	Parasitic on floodplain trees	🟩
Lythrum hyssopifolia	Lythraceae	Floodplain	🟩
Lythrum salicaria	Lythraceae	Amphibious fluctuation tolerator–emergent	🟩
Maireana brevifolia	Chenopodiaceae	Terrestrial dry	
Maireana enchylaenoides	Chenopodiaceae	Terrestrial dry	
Maireana macrocarpa	Chenopodiaceae	Terrestrial dry	
Maireana microcarpa	Chenopodiaceae	Floodplain	🟩
Maireana oppositfolia	Chenopodiaceae	Terrestrial dry	
Maireana sclerolaenoides	Chenopodiaceae	Terrestrial dry	
Maireana sedifolia	Chenopodiaceae	Terrestrial dry	
*Malva parvifolium**	Malvaceae	Terrestrial dry	
*Marrubium vulgare**	Lamiaceae	Terrestrial dry	
Marsilea angustifolia	Marsileaceae	Amphibious fluctuation responder–plastic	🟩
Marsilea drummondii	Marsileaceae	Amphibious fluctuation responder–plastic	🟩
*Medicago minima**	Fabaceae	Terrestrial dry	
*Medicago polymorpha var. polymorpha**	Fabaceae	Terrestrial dry	
*Medicago polymorpha**	Fabaceae	Terrestrial dry	
*Medicago truncatula**	Fabaceae	Terrestrial dry	
*Melilotus alba**	Fabaceae	Terrestrial dry	
*Melilotus indica**	Fabaceae	Terrestrial dry	
Mentha australis	Lamiaceae	Amphibious fluctuation responder–plastic	🟩
*Mentha piperita**	Lamiaceae	Amphibious fluctuation responder–plastic	
Mentha sp.*	Lamiaceae	Amphibious fluctuation responder–plastic	
*Mesembryanthemum crystallinum**	Aizoaceae	Terrestrial dry	
*Mesembryanthemum nodiflorum**	Aizoaceae	Terrestrial dry	
Mimulus repens	Scrophulariaceae	Amphibious fluctuation responder–plastic	🟩
Minuria sp.	Asteraceae	Terrestrial dry	
Mollugo cerviana	Aizoaceae	Floodplain	🟩
Muehlenbeckia gunnii	Polygonaceae	Terrestrial dry	
Myoporum parvifolium#	Myoporaceae	Floodplain	🟩
Myosurus minimus var. *australis*	Ranunculaceae	Floodplain	🟩
Myriocephalus stuartii	Asteraceae	Floodplain	🟩
Myriophyllum caput–medusae	Haloragaceae	Amphibious fluctuation responder–plastic	🟩

Species	**Family**	**Functional Group**	**Native species associated with aquatic or floodplain ecosystems**
Myriophyllum crispatum	Haloragaceae	Amphibious fluctuation responder–plastic	🟩
Myriophyllum papulosum#	Haloragaceae	Amphibious fluctuation responder–plastic	🟩
Myriophyllum salsugineum	Haloragaceae	Amphibious fluctuation responder–plastic	🟩
Myriophyllum simulans	Haloragaceae	Amphibious fluctuation responder–plastic	🟩
Myriophyllum verucossum	Haloragaceae	Amphibious fluctuation responder–plastic	🟩
Neogunnia septifraga	Aizoaceae	Floodplain	🟩
*Nicotiana glauca**	Solanaceae	Terrestrial dry	
Nicotiana goodspeedii	Solanaceae	Terrestrial dry	
Nicotiana velutina	Solanaceae	Terrestrial dry	
Nitella sp.	Characeae		🟩
*Nothoscordum inodorata**	Liliaceae	Terrestrial dry	
Nymphaea sp.*	Nymphaceae	Amphibious fluctuation responder–plastic	
Oenothera sp.	Onagraceae	Terrestrial dry	
Osteocarpum acropterum var. *acropterum*	Chenopodiaceae	Floodplain	🟩
Ottelia ovalifolia#	Hydrocharitaceae	Amphibious fluctuation responder–plastic	🟩
*Oxalis pes–caprae**	Oxalidaceae	Terrestrial dry	
Pachycornia arbuscula	Chenopodiaceae	Terrestrial dry	
Pachycornia triandra	Chenopodiaceae	Terrestrial dry	
*Parapholis incurva**	Poaceae	Terrestrial damp	
Paspalidium jubiflorum	Poaceae	Terrestrial damp	
*Paspalum dilatatum**	Poaceae	Terrestrial damp	
Paspalum distichum	Poaceae	Amphibious fluctuation tolerator–emergent	🟩
*Paspalum vaginatum**	Poaceae	Terrestrial damp	
*Pennisetum clandestinum**	Poaceae	Terrestrial dry	
Persicaria decipiens	Polygonaceae	Amphibious fluctuation responder–plastic	🟩
Persicaria lapathifolium	Polygonaceae	Amphibious fluctuation responder–plastic	🟩
*Phalaris arundinacea**	Poaceae	Amphibious fluctuation tolerator–emergent	
*Phalaris minor**	Poaceae	Amphibious fluctuation tolerator–emergent	
*Phalaris paradoxa**	Poaceae	Amphibious fluctuation tolerator–emergent	
Phragmites australis	Poaceae	Emergent	🟩
*Phyla canescens**	Verbeneaceae	Amphibious fluctuation tolerator low growing	
Phyllanthus lacunaris	Euphorbiaceae	Floodplain	🟩
Picris hieracoides var. *hieracoides**	Asteraceae	Terrestrial dry	
Pimelea glauca	Thymeleaeceae	Terrestrial dry	
Pimelea penicillaris#	Thymeleaeceae	Terrestrial dry	
Plantago coronopus ssp. *coronopus**	Plantaginaceae	Terrestrial dry	
Plantago cunninghamii	Plantaginaceae	Floodplain	🟩
*Plantago lanceolata**	Plantaginaceae	Terrestrial dry	
*Plantago major**	Plantaginaceae	Terrestrial dry	
Plantago turrifera	Plantaginaceae	Floodplain	🟩
*Poa annua**	Poaceae	Terrestrial damp	
Poa labillardieri var. *labillardieri*	Poaceae	Terrestrial dry	
Poa poiformis	Poaceae	Terrestrial dry	
*Polygonum aviculare**	Polygonaceae	Terrestrial dry	
Polygonum plebeium	Polygonaceae	Floodplain	🟩
*Polypogon maritimus**	Poaceae	Amphibious fluctuation tolerator–emergent	
*Polypogon monspeliensis**	Poaceae	Amphibious fluctuation tolerator–emergent	
*Polypogon viridis**	Poaceae	Amphibious fluctuation tolerator–emergent	
Portulaca sp.	Portulaceae	Terrestrial dry	
Potamogeton crispus	Potamogetonaceae	Submergent k–	🟩

Species	Family	Functional Group	Native species associated with aquatic or floodplain ecosystems
		selected	🟩
Potamogeton pectinatus	Potamogetonaceae	Submergent k–selected	🟩
Potamogeton tricarinatus	Potamogetonaceae	Amphibious fluctuation responder–plastic	🟩
Helichrysum luteo–album	Asteraceae	Floodplain	🟩
*Psilocaulon tenue**	Aizoaceae	Terrestrial dry	
Ptilotus exaltatus	Amaranthaceae	Terrestrial dry	
Ptilotus obovatus	Amaranthaceae	Terrestrial dry	
*Puccinellia distans**	Poaceae	Terrestrial dry	
Puccinellia fasciculata	Poaceae	Amphibious fluctuation tolerator–emergent	🟩
Puccinellia stricta var. *perlaxa*	Poaceae	Terrestrial dry	
Puccinellia stricta var. *stricta*	Poaceae	Terrestrial dry	
Pycnosorus globosus	Asteraceae	Floodplain	🟩
Ranunculus amphitrichus	Ranunculaceae	Amphibious fluctuation responder–plastic	🟩
Ranunculus rivularis	Ranunculaceae	Amphibious fluctuation responder–plastic	🟩
*Ranunculus sceleratus***	Ranunculaceae	Amphibious fluctuation responder–plastic	
*Ranunculus trilobus**	Ranunculaceae	Amphibious fluctuation responder–plastic	
Rhagodia candolleana ssp. *candolleana*	Chenopodiaceae	Terrestrial dry	
Rhagodia parabolica	Chenopodiaceae	Terrestrial dry	
Rhagodia spinescens	Chenopodiaceae	Terrestrial dry	
Rhodanthe pygmaeum	Asteraceae	Floodplain	🟩
Riechardia tingitana	Asteraceae	Terrestrial dry	
Romulea rosea	Iridaceae	Terrestrial dry	
Rorippa eustylis	Brassicaceae	Floodplain	🟩
Rorippa islandica	Brassicaceae	Floodplain	🟩
*Rorippa nasturtium–aquaticum**	Brassicaceae	Amphibious fluctuation responder–plastic	
*Rorippa palustris**	Brassicaceae	Floodplain	
*Rostraria cristata**	Poaceae	Terrestrial dry	
Rumex bidens	Polygonaceae	Amphibious fluctuation responder–plastic	🟩
*Rumex conglomeratus**	Polygonaceae	Amphibious fluctuation responder–plastic	
*Rumex crispus**	Polygonaceae	Amphibious fluctuation responder–plastic	
Rumex pulcher ssp. *pulcher**	Polygonaceae	Amphibious fluctuation responder–plastic	
Ruppia megacarpa	Potamogetonaceae	Submergent k–selected	🟩
Ruppia polycarpa	Potamogetonaceae	Submergent r–selected	🟩
Ruppia tuberosa	Potamogetonaceae	Submergent r–selected	🟩
*Sagina apetala**	Caryophyllaceae	Terrestrial dry	
*Sagina maritima**	Caryophyllaceae	Terrestrial dry	
*Salix nigrum***	Salicaceae	Emergent	
*Salix babylonica**	Salicaceae	Emergent	
*Salix fragilis***	Salicaceae	Emergent	
Salix matsudana "Tortuosa"**	Salicaceae	Emergent	
*Salix nigrum***	Salicaceae	Emergent	
Salsola kali var. *kali*	Chenopodiaceae	Terrestrial dry	
Samolus repens	Primulaceae	Floodplain	🟩
Sarcocornia blackiana	Chenopodiaceae	Amphibious fluctuation tolerator–emergent	🟩
Sarcocornia quinqueflora	Chenopodiaceae	Amphibious fluctuation tolerator–emergent	🟩
*Scabiosa atropurpurea**	Dipsacaceae	Terrestrial dry	
*Schismus barbatus**	Poaceae	Terrestrial dry	
Schoenoplectus pungens	Cyperaceae	Amphibious fluctuation tolerator–emergent	🟩
Schoenoplectus validus	Cyperaceae	Emergent	🟩
Schoenus apogon	Cyperaceae	Floodplain	🟩

Species	**Family**	**Functional Group**	**Native species associated with aquatic or floodplain ecosystems**
Scleroblitum atriplicinum	Chenopodiaceae	Floodplain	🟩
Sclerolaena blackiana	Chenopodiaceae	Terrestrial dry	
Sclerolaena brachyptera	Chenopodiaceae	Terrestrial dry	
Sclerolaena decurrens	Chenopodiaceae	Terrestrial dry	
Sclerolaena diacantha	Chenopodiaceae	Terrestrial dry	
Sclerolaena muricata var. *muricata*	Chenopodiaceae	Terrestrial dry	
Sclerolaena muricata var. *villosa*	Chenopodiaceae	Terrestrial dry	
Sclerolaena obliquicuspis	Chenopodiaceae	Terrestrial dry	
Sclerolaena paradoxa	Chenopodiaceae	Terrestrial dry	
Sclerolaena stelligera	Chenopodiaceae	Terrestrial dry	
Sclerolaena tricuspis	Chenopodiaceae	Terrestrial dry	
Sclerostegia arbuscula	Chenopodiaceae	Floodplain	🟩
Selliera radicans	Goodeniaceae	Amphibious fluctuation responder–plastic	🟩
Senecio cunninghamii	Asteraceae	Floodplain	🟩
Senecio glossanthus	Asteraceae	Terrestrial dry	
Senecio pterophorus var. *pterophorus**	Asteraceae	Terrestrial dry	
Senecio runcinifolius	Asteraceae	Floodplain	🟩
Sida ammophila	Malvaceae	Terrestrial dry	
*Silene apetala**	Caryophyllaceae	Floodplain	
*Silene nocturna**	Caryophyllaceae	Floodplain	
*Sinapis alba**	Brassicaceae	Floodplain	
*Sisymbrium erysimoides**	Brassicaceae	Floodplain	
*Sisymbrium irio**	Brassicaceae	Floodplain	
Solanum lacunarium	Solanaceae	Floodplain	🟩
*Solanum nigrum**	Solanaceae	Terrestrial dry	
Solanum oliganthum	Solanaceae	Floodplain	🟩
Sonchus asper ssp. *glaucescens**	Asteraceae	Floodplain	
*Sonchus hydrophilus**	Asteraceae	Floodplain	
*Sonchus oleraceus**	Asteraceae	Terrestrial dry	
*Spergularia marina**	Caryophyllaceae	Floodplain	
Spirodela punctata	Lemnaceae	Floating	🟩
Sporobolus mitchelli	Poaceae	Floodplain	🟩
Sporobolus virginicus	Poaceae	Terrestrial damp	
Stemodia florulenta	Scrophulariaceae	Floodplain	🟩
Stipa drummondii	Poaceae	Terrestrial dry	
Stipa nitida	Poaceae	Terrestrial dry	
Stipa stipoides	Poaceae	Terrestrial dry	
Suaeda australis	Chenopodiaceae	Amphibious fluctuation tolerator–emergent	🟩
Swainsona greyana	Fabaceae	Floodplain	🟩
Swainsona swainsonoides	Fabaceae	Floodplain	🟩
*Silybum marianum**	Asteraceae	Terrestrial damp	
*Tamarix aphylla**	Tamaricaeae	Terrestrial dry	
*Taraxacum officinale**	Asteraceae	Terrestrial damp	
Tecticornia halocnemoides	Chenopodiaceae	Amphibious fluctuation tolerator–emergent	🟩
Tecticornia indica ssp. *leiostachya*	Chenopodiaceae	Amphibious fluctuation tolerator–emergent	🟩
Tecticornia pergranulata ssp. *pergranulata*	Chenopodiaceae	Amphibious fluctuation tolerator–emergent	🟩
Tetragonia eremaea	Aizoaceae	Floodplain	🟩
Tetragonia implexicoma	Aizoaceae	Floodplain	🟩
Tetragonia tetragonioides	Aizoaceae	Floodplain	🟩
Teucruim racemosum	Lamiaceae	Terrestrial dry	
Threlkeldia diffusa	Chenopodiaceae	Terrestrial dry	
Trachymene cyanopetula	Apiaceae	Floodplain	🟩
Trifolium arvense var. *arvense**	Fabaceae	Terrestrial dry	
*Trifolium repens**	Fabaceae	Terrestrial dry	
Triglochin hexagonum	Juncaginaceae	Emergent	🟩
Triglochin procerum	Juncaginaceae	Emergent	🟩
Triglochin striatum	Juncaginaceae	Amphibious fluctuation tolerator–emergent	🟩

Species	Family	Functional Group	Native species associated with aquatic or floodplain ecosystems
Typha domingensis	Typhaceae	Emergent	🟩
Typha orientalis	Typhaceae	Emergent	🟩
*Urospermum picroides**	Asteraceae	Terrestrial dry	
Urtica incisa	Urticaceae	Floodplain	🟩
*Urtica urens**	Urticaceae	Terrestrial damp	
Vallisneria australis	Hydrocharitaceae	Submergent k–selected	🟩
*Verbascum virgatum**	Scrophulariaceae	Terrestrial dry	
*Vicia sativa**	Fabaceae	Terrestrial dry	
Viminaria juncea	Fabaceae	Terrestrial dry	
Vittadinia australasica	Asteraceae	Terrestrial dry	
Vittadinia cervicularis	Asteraceae	Terrestrial dry	
Vittadinia cuneata var. *cuneata*	Asteraceae	Terrestrial dry	
Vittadinia gracilis	Asteraceae	Terrestrial dry	
Vulpia fasciculata	Poaceae	Terrestrial dry	
Vulpia muralis	Poaceae	Terrestrial dry	
Wahlenbergia fluminalis	Campanulaceae	Floodplain	🟩
Wilsonia rotundifolia	Convolulaceae	Floodplain	🟩
*Xanthium californicum**	Asteraceae	Floodplain	
*Xanthium occidentale**	Asteraceae	Floodplain	
Zannichellia palustris#	Zannichelliaceae	Submergent k–selected	🟩