Australian Government

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MURRAY FUTURES

Riverine Recovery

Riverine Recovery

Monitoring and Evaluation Program -

Conceptual understanding of the ecological response to water level manipulation

WATER FOR GOOD

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Further information on the Riverine Recovery Project:

Environmental Water Management Program, Department of Environment, Water and Natural Resources http://www.murrayfutures.sa.gov.au/riverine.php.

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Additional information and understanding presented also relies upon the accumulated knowledge of researchers and managers of the River Murray in South Australia over a long period of time.

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Riverine Recovery Project

Background

Murray Futures is a ten-year program aimed at restoring the River Murray in South Australia. It is funded principally by the Australian Government through the *Water for the Future* program. The *Riverine Recovery Project* (RRP) is a major element of *Murray Futures*. The RRP has a budget of $100 million to provide targeted delivery of environmental water to help restore ecological function to the River Murray floodplain. It also aims to facilitate the adaptation of the River Murray system to current and projected impacts of drought and climate change (DWLBC 2008). The project will build upon years of work by Government and local communities and support and extend programs that are underway in South Australia.

The study area for the RRP is the River Murray from the South Australian border to Wellington, extending laterally to the area inundated by the 1956 flood (Figure 1).

The RRP has three objectives which are linked to the broad objectives of *Murray Futures*:

- Improve the health of wetlands, floodplains and the river (i.e. establish measurable and scalable environmental, social and economic goals and targets for riverine health and establish an adaptive management system for selected wetlands and floodplains to achieve these goals and targets).

- Save water for the environment and be climate-change ready (i.e. establish a water use system that will allow use of available water for best environmental outcomes in wetlands and on floodplains).

- Give security to regional communities (i.e. relocate pumps from wetlands to the river channel to provide better water quality and secure access for irrigators currently dependent on a stable water level in the selected wetlands) (DWLBC 2008).

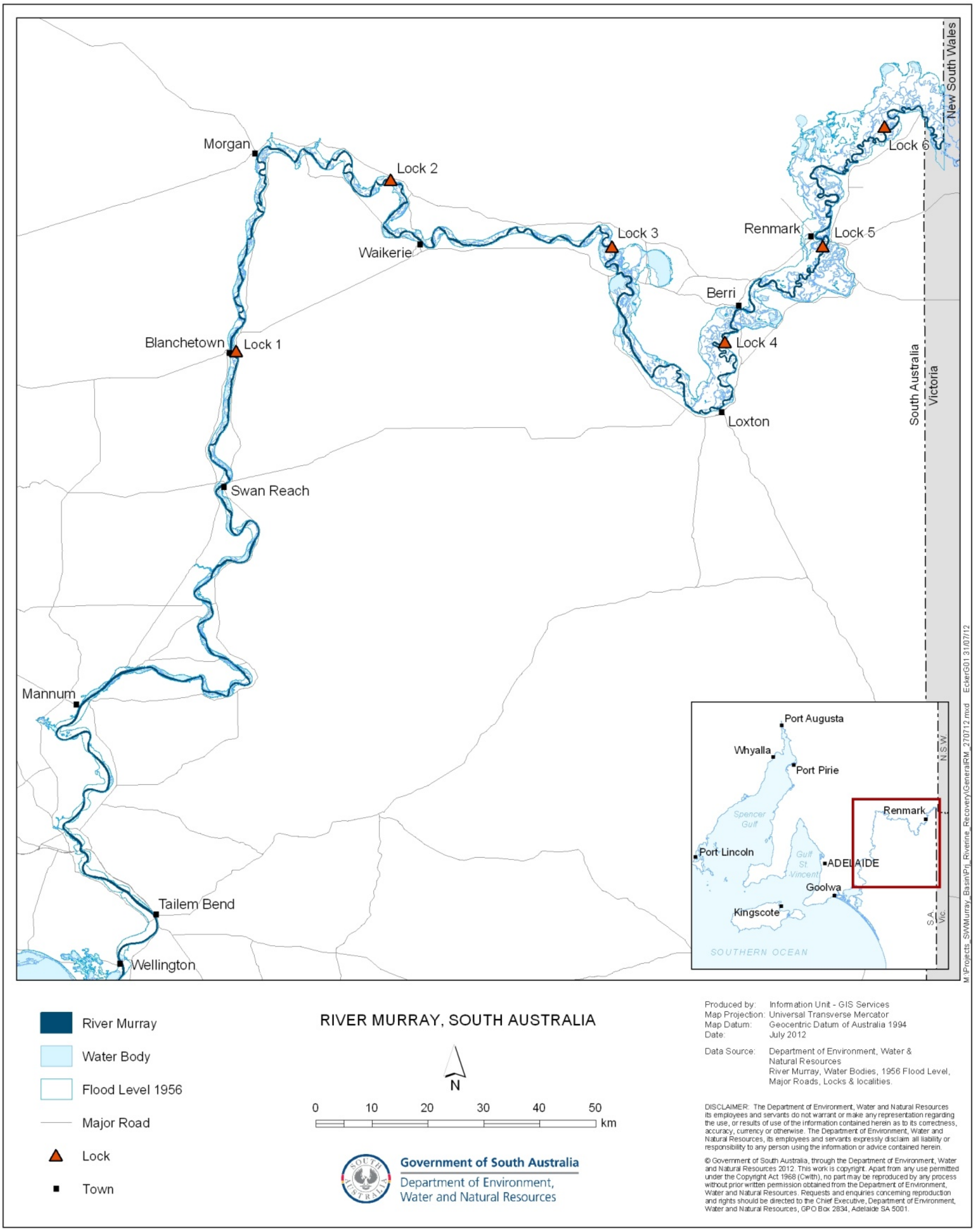

Figure 1: Map of the study region in South Australia showing the River Murray and area inundated by the 1956 flood. Major towns and the locations of the locks on the Lower River Murray area also shown

Ecological goals of the Riverine Recovery Project

Water regime is the timing, duration, frequency, extent, depth of inundation and the variability of water presence (Boulton and Brock 1999). Prior to river regulation a gradient in wetland types defined by the water regime (i.e. a gradient from temporary to permanently inundated wetlands) would have existed on the Lower River Murray. Since water regime is a principal driver of wetland ecology, regulation has had major impacts on these wetlands, essentially 'de-synchronising' fundamental links between hydrological cycles and the ecology of wetland flora and fauna (Kingsford 2000).

The maintenance of stable weir pool levels due to river regulation has resulted in many wetlands now tending towards either of the two hydrological extremes (Figure 2). Wetlands with sill levels above normal weir pool level suffer from extended periods of dryness. The exceptions are those wetlands below Lock and Weir 1 that experience regular water level variations of approximately 0.6 m due to wind driven movement and evaporation in this reach. Wetlands with sill levels below normal weir pool level tend to be permanently inundated with little water level fluctuation (75% of wetland area between the South Australian border and Wellington; Jones and Miles 2009; Pressey 1986). It is thought that static water regimes push the river and permanent wetlands towards alternate stable states. Overall, the maintenance of stable water levels has led to relatively simple and narrow vegetation assemblages that support communities of generalist consumers, with a demise in wetland specialists (e.g. some threatened fish species).

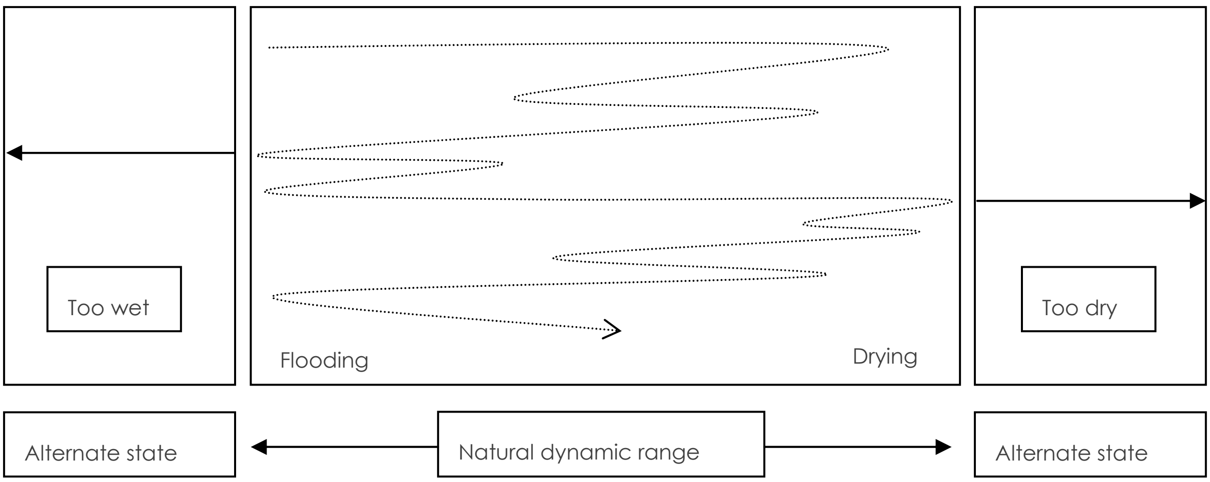

Figure 2: Conceptual model of the natural dynamic water regime of wetlands and the effects of river regulation, causing transitions to either of two alternate states (adapted from Wallace et al. (2011))

RRP aims to restore the gradient in hydrological regimes that existed prior to river regulation by creating greater water level variability in individual wetlands and at the reach scale. This will be achieved through the operation of wetland regulatory structures and the manipulation of weir pool levels. Weir pool manipulations affect a whole reach by changing the height of a weir (raising or lowering) and can significantly alter river, wetland and floodplain water levels compared to pool level. Wetland flow regulators can be closed to lower water levels (through evaporation) below pool level and opened to connect the wetland to the river and stabilise water levels at pool level.

Whilst the manipulation of wetland water levels and weir pool water levels operate at different spatial scales, the reasons for manipulation are similar. In addition, these levers can operate together or individually. Water levels will be varied to create areas that are inundated at varying frequencies and durations and in doing so create a diversity of habitats for wetland and floodplain biota. Figure 3 shows the broad ecological goals of the RRP. Manipulation of water regime will directly and indirectly affect water and sediment quality, vegetation and consumer groups, including frogs, fish and birds. Although all the components and processes depicted in Figure 3 are driven by water regime, they are also inter-connected and many feedback loops may occur. For example, there are many feedbacks that control the switch between a turbid, phytoplankton-dominated state and the clear-water vegetation-dominated state (Turbidity section).

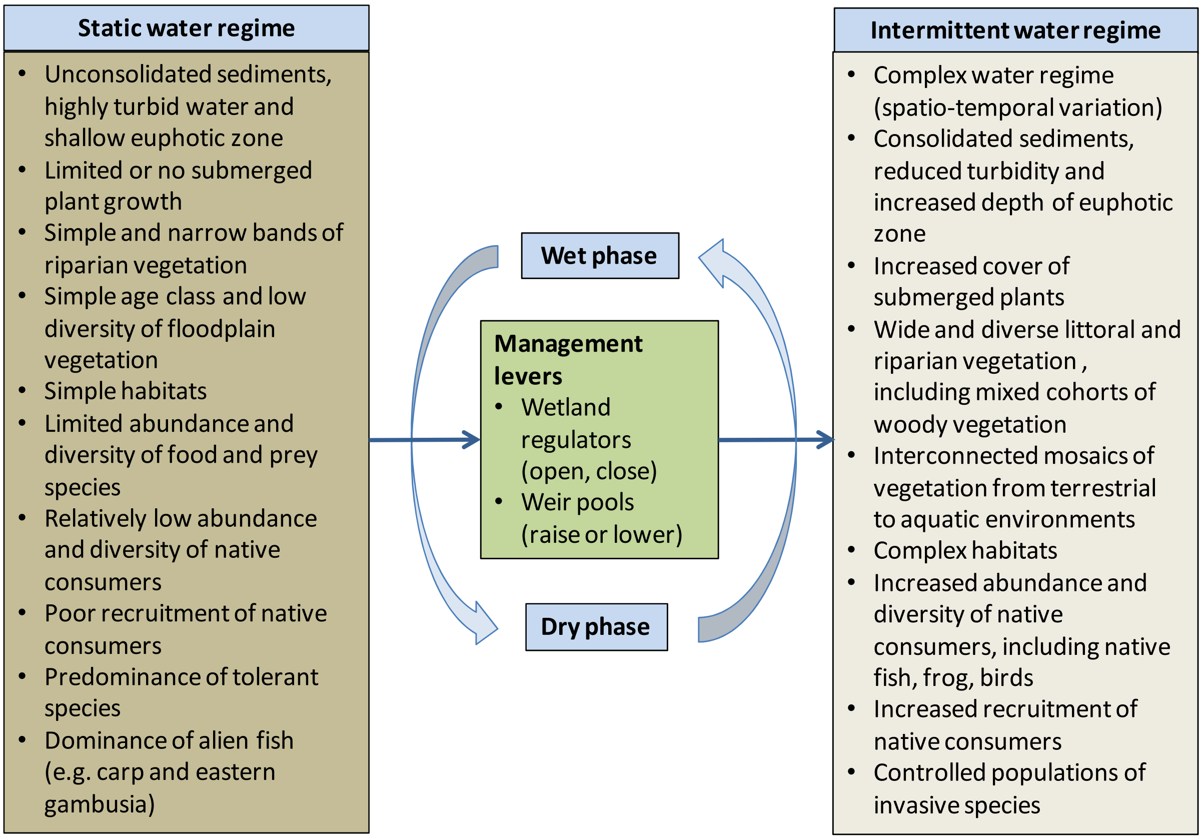

Figure 3: Broad goals of water level manipulation of the Riverine Recovery Project

Monitoring and evaluation program development

This program builds on previous work that has identified the processes influencing the ecological condition of the River Murray system. It also builds on previous work that has established the basis for monitoring and management of the River Murray system (e.g. Wilkinson *et al.* 2007a, 2007b).

Wilkinson *et al.* (2007a) identified four components that are essential to developing an effective monitoring and evaluation program:

1. Rationale and priorities: provides justification for the development of the program; determines and categorises objectives; identifies the physical and biological nature of water dependent ecosystems; defines and tests the objectives and targets; and documents the assumptions made.

2. Conceptual understanding: developed either as conceptual diagrams (showing the major ecosystem components and the influences on conditions at the landscape scale), a stressor model (portraying key stress response relationships affecting the system) and/or a state-and-transition model (for systems which progress from one condition through various stages and back to the initial condition).

3. Monitoring program: identifies the scope of the monitoring program, link the monitoring needs with existing programs and recommend the content of the monitoring program. e.g. indicators and measures of progress towards environmental goals, frequency of data collection and indicators.

4. Implementation and assessment: implementation of the monitoring program and review of information collected to allow adaptive management and improve the project outcomes. This involves evaluating and assessing data collected within the project, reviewing original objectives to determine the effectiveness of the project and reporting on findings, lessons learned and recommendations for improvement.

Background work that has addressed some of the requirements listed above, includes:

- Souter (2009) which addressed much of component 2 by developing conceptual and stressor models and stommel diagrams (indicating temporal and spatial scales of response) for River Murray wetlands and floodplains. This allowed the current work to focus on state-and-transition characteristics of wetlands and weir pools in response to the implementation of a managed water regime (see DEWNR 2012a).

- The Monitoring and Management Framework (Aquaterra 2010) facilitated the assessment of management actions against the RRP objectives and targets. This will enable ongoing review of monitoring and improvement of management actions. It has informed the development of this program by establishing linkages of the goals and objectives of the RRP and other projects, including:

    - River Health Project (Bull and Sheldon 2009)
    - Wetland Classification and Prioritisation Project (Jones and Miles 2009)
    - Wetland Management Guidelines (DWLBC 2004)
    - Environmental Water Requirements Project (Ecological Associates 2010)
    - Conceptual Models Project (Souter 2009)
    - Management Action Database (MAD) (Gunko 2010).

Between 2003 and 2007 baseline surveys were conducted at over 60 Lower River Murray wetlands to inform adaptive management (SKM 2006a, 2006b). Of these sites, 22 had actively managed water regimes. Data were collected on a range of physical and biological and wetland characteristics (SKM 2006a) including:

- wetland bathymetry
- groundwater levels and salinity
- water quality
- vegetation composition and zonation
- fish, water-bird and frog communities.

Report description

Together, this report and its companion report (DEWNR 2012) focuses on the underlying conceptual understanding and technical aspects of data collection, analysis and interpretation of the RRP. This report details the conceptual understanding of how it is anticipated that manipulating water levels by the RRP will result in the desired ecological response in the Lower River Murray. This conceptual understanding was developed through a combination of literature, observations and expert opinion. It forms a basis upon which the monitoring and evaluation program will provide evidence for the actual response to water level manipulations.

Current understanding is synthesised into conceptual models, illustrating the characteristics of wetlands expected under static and variable water regimes. Not all components of the ecosystem are discussed. Selected components are vegetation, waterbirds, frogs and fish communities on the basis that they: are able to inform management; allow comparisons to previous baseline surveys (Monitoring and evaluation program development section); and possess characteristics of suitable ecological indicators (See DEWNR 2012). Considerable effort is dedicated to conceptualising the vegetation response since it is anticipated that vegetation will respond directly to water level manipulations and the vegetation response is critical to the response of all consumer indicators (fish, water-birds and frogs). Sediment and water quality are identified as factors that are most likely to limit the desired ecological response to water level manipulations and so information on the likely response of these factors is also provided.

The conceptual understanding described here forms a platform for planning water level manipulations and monitoring and evaluation. This conceptual understanding will also allow testable hypotheses to be developed when preparing adaptable management plans for individual wetlands. These hypotheses can be verified through the monitoring and evaluation program (See DEWNR 2012). DEWNR (2012) describes the design of the monitoring and evaluation program, the selection of indicators for the monitoring program, monitoring methods for selected indicators and guidance on how to evaluate and interpret data that is collected.

Monitoring and management framework

This program is broadly consistent with the principles of the monitoring, evaluation, reporting and improvement (MERI) framework of the Australian Government (Australian Government 2009), which sets out the logic by which program progress will be reported. The program is based on adaptive management principles. Management goals are based on the identification of a managed asset's current condition (step 1); a conceptual understanding of how the system functions (step 2); and the use of this information to determine a desired state (step 3) (Figure 4). Management interventions are described as a series of targets (step 4) achieved through water level manipulation (step 5), which will be evaluated through data collection and analysis (step 6).

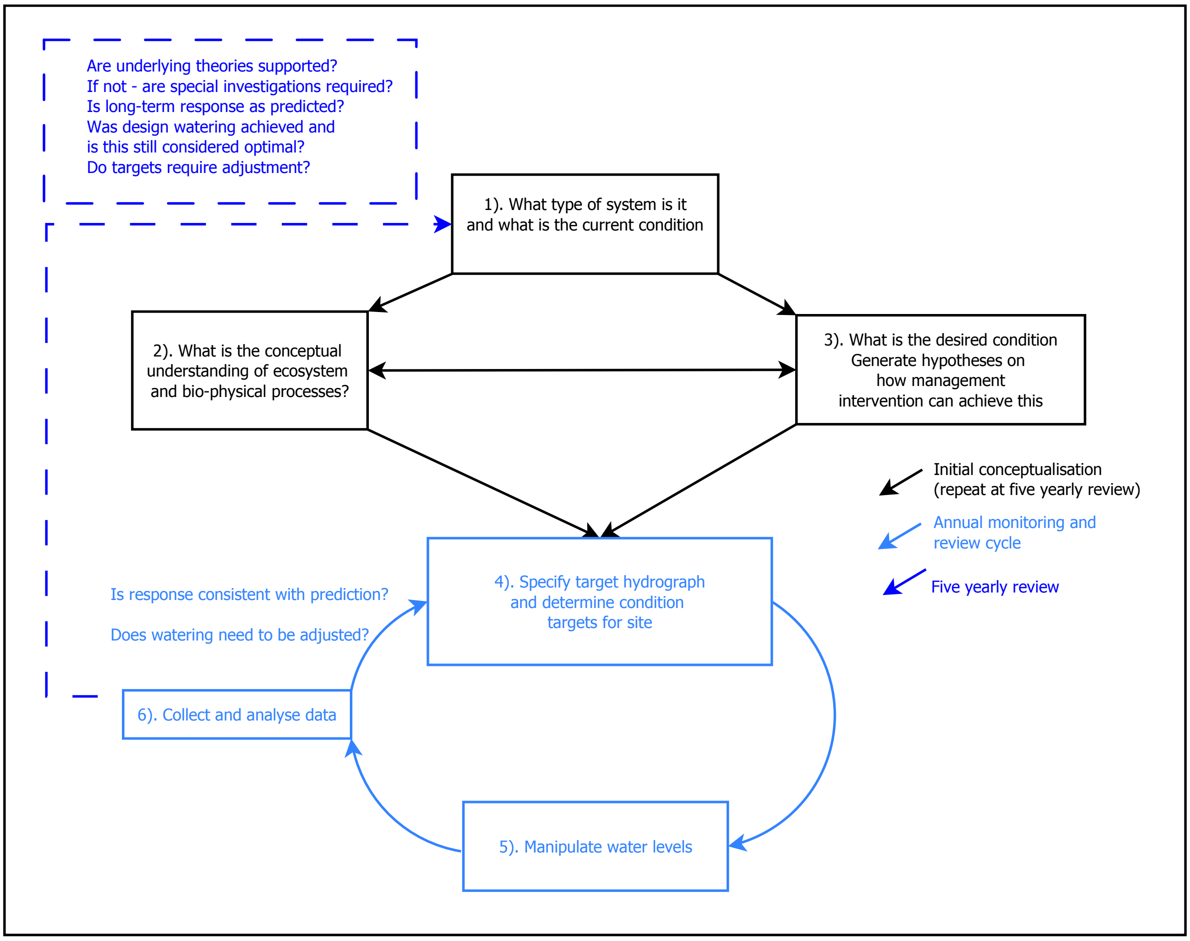

Figure 4: Monitoring and management program structure

For wetlands that are currently actively managed, step 1 is addressed through baseline surveys (e.g. SKM 2006a) and pre-intervention rapid assessments (e.g. Aldridge *et al.* 2012b). For other wetlands and weir pool manipulations, the baseline condition of the monitoring sites needs to be established prior to intervention. Much of the conceptual understanding required for step 2 has been established through analysis of a large body of prior work, as described in the following sections.

Step 3 identifies the targets of the assets. Targets must consider the expected short-term and long-term changes, as wetlands change from the current to the desired condition. These expected outcomes (targets) need to be specific, measurable, achievable, realistic and time bound (SMART) to enable monitoring to track progress (Wilkinson *et al*. 2007a). The targets for this project were largely governed by those of the water management plans for individual wetlands. A review of existing management plans revealed several consistent environmental targets, such as increasing the diversity and condition of vegetation. This program has adapted the targets so that they also align with the principles of SMART targets and so that they can be assessed at wetland and landscape scales (DEWNR 2012a).

Once targets are identified, the optimal hydrograph to achieve these targets is identified (step 4). These are underpinned with specific testable hypotheses (see DEWNR 2012a) that describe the processes through which it is anticipated that targets will be achieved. These hypotheses provide a basis for monitoring and statistical analyses. Information requirements for the RRP relate to both trend (change over time) and status (current condition) of managed and unmanaged sites.

Once the desired state for the system is determined, each managed wetland or manipulated weir pool will be subject to a specified wetting and drying cycle (or raising and lowering protocol) (step 5). Monitoring data will then be collected and analysed (step 6) to measure progress along the hypothesised trajectory of change and establish the statistical inference on which the success of interventions for achieving targets is judged. Specifically, monitoring will be used to:

- assess progress towards targets and test hypotheses
- assess benefits and risks as part of the adaptive management process
- identify and manage drivers of change
- improve operations and prevent long-term damage
- justify investment to achieve objectives.

The management plans for individual wetlands are written for a five-year cycle, after which they will be reviewed. Monitoring and adaptive management reviews will occur more frequently than every five years, ideally before and after every intervention. These reviews will be conducted to assess progress and refine the conceptual understanding, targets and monitoring effort. In addition to annual consideration of site-level progress, adaptive management requires periodic review of the entire program at the landscape scale. Meta-analysis of monitoring data collected from all the RRP sites can improve our understanding of riverine processes at the landscape scale (the scale of the RRP program, Figure 1). Overall, the monitoring program needs to provide answers to the following questions, which have been fundamental in determining the technical aspects of program design (DEWNR 2012):

- Did intervention achieve what was anticipated?
- Are we on track to achieve our targets?
- Has anything unexpected arisen?
- Is it necessary to change our targets, monitoring, conceptual understanding or management approach?

Vegetation response to water level manipulation

Background

Water level changes in a wetland interact with elevation to determine the water regime a plant may experience in its habitat (e.g. permanently aquatic, intermittent floodplain or dry terrestrial), with wetter habitats occurring further down the elevation gradient and *vice versa*. Other drivers such as turbidity and nutrient levels also strongly affect plant dynamics (Banach *et al.* 2009; Blanch *et al.* 1999a, 1999b, 2000; Deegan *et al.* 2007; Grillas 1990; Keddy and Ellis 1985; Nicol and Ganf 2000; Nicol *et al.* 2003; Nielsen and Chick 1997; Rea and Ganf 1994a, 1994b, 1994c, 1994d).

Due to the relatively static water levels in wetlands of the River Murray that have resulted from regulation, plants that prefer, or require, a given water regime have a narrow elevation band within which that water regime is provided. When coupled with high turbidity, the static water regime results in a highly constrained light environment and in turn highly constrained vegetation communities. Blanch *et al*. (1998) showed that turbid Darling River waters (up to 504 NTU) prevented growth of *Vallisneria americana* in only 0.5 m of water, whereas *V. americana* could grow in clear water up to 2 m deep. Therefore, bands of vegetation types are often narrow and relatively distinct, with little overlap between plants from different functional groups (Table 1). The static water regime also favours certain plants that can rapidly spread through vegetative growth (e.g. *Typha domingensis* and *Phragmites australis*) and outcompete plants that have life cycles more suited to variable water regimes (e.g. *Persicaria* spp., *Ludwigia peploides*). Therefore the narrow vegetation bands also tend to be less diverse in species, life cycle stage and growth form. This is supported by observations of increased vegetation diversity below Lock 1 where wind seiching drives fluctuations in water level within wetlands held at pool level (J. Nicol, South Australian Research and Development Institute, pers. comm.).

The impacts of a static water regime can be seen in the riparian vegetation communities. For example, *Eucalyptus camaldulensis* (river red gum) and *E. largiflorens* (black box) require flooding (Close 1990), or perhaps intermittent inundation, in order to successfully recruit new plants into the population. River red gum and black box communities tend to have age structures that are dominated by trees of similar ages (cohorts) that correspond to unregulated floods (e.g. six floods between 1880 and 1970s determined by Dexter (1967, 1978); also see Breen *et al.* (1988)).

Wetland plant types and characteristics

Functional classification: Wetland plants are typically categorised on the basis of their habit (form), water dependencies and/or physico-chemical tolerances. These plant attributes are combined to yield a classification of plants into functional groups (Table 1 and Figure 5). The water regime required for the different plant functional groups can be used to explain the patterns observed in a wetland.

Table 1: Plant functional groups with examples of water dependent taxa found in River Murray wetlands. Adapted from Brock and Casanova 1997, Nicol *et al.* 2010 and Casanova 2011

Functional group	Water regime preference	River Murray examples
Terrestrial dry	Will not tolerate inundation and tolerates low soil moisture for extended periods.	*Atriplex vesicaria, Sclerolaena divaricata, Frankenia pauciflora*
Terrestrial damp	Germinate/establish on saturated or damp ground, cannot tolerate flooding in the vegetative state. Require high soil moisture throughout their life cycle.	*Conyza bonariensis, Chenopodium glaucum, Distichlis distichophylla*
Floodplain	Temporary inundation, plants germinate on newly exposed soil after flooding but not in response to rainfall.	*Epaltes australis, Centipeda minima, Glinus lotoides*
Amphibious fluctuation tolerator-emergent	Survive in saturated soil or shallow water but require most of their photosynthetic parts to remain above the water. They tolerate fluctuations in the depth of water, as well as water presence. They need water to be present for c. 8–10 months of the year, and the dry time to be in the cooler months of the year	*Juncus usitatus, Cyperus gymnocaulos, Halosarcia pergranulata*
Amphibious fluctuation tolerator-woody	Require water to be present in the root zone all year round, but will germinate in shallow water or on a drying profile.	*Eucalyptus camaldulensis, Muehlenbeckia florulenta, Acacia stenophylla*
Amphibious fluctuation tolerator-low growing	Germinate either on saturated soil or under water and grow totally submerged, as long as they are exposed to air by the time they start to flower and set seed. They require shallow flooding for c. 3 months.	*Isolepis hookeriana, Mimulus repens, Crassula sieberiana*
Amphibious fluctuation responder-plastic	Similar zone to the above group, except that they have a morphological response to water level changes such as rapid shoot elongation or a change in leaf type.	*Ludwigia peploides, Limosella australis, Myriophyllum verrucosum*
Amphibious fluctuation responder–floating	Grow underwater or float on the surface of the water or have floating leaves. They require the year-round presence of free water, but many can survive and complete their life cycle stranded on mud.	*Azolla* spp., *Lemna* spp., *Wolffia* spp.
Emergent	Require permanent water in the root zone, but remain emergent	*Typha* spp., *Phragmites australis, Triglochin procerum, Schoenoplectus validus*
Submergent k-selected	Require permanent inundation. Many have asexual reproduction (fragmentation, rhizomes and turions).	*Vallisneria spiralis, Potamogeton crispus, Zanichellia palustris*
Submergent r-selected	Inhabit temporary waters with their habitats flooded from once a year to once a decade, to a depth >0.1 m. Many require drying to stimulate high germination percentages, and they frequently complete their life cycle quickly and die off naturally. They persist via a dormant, long-lived bank of seeds or spores in the soil.	*Ruppia tuberosa, Chara* spp., *Lepilaena* spp.
Floating	Free-floating with either the whole plant or just the leaf tissue floating on the surface. Highly susceptible to wind and water movement and are readily dispersed between hydrologically connected habitats.	*Azolla filiculoides*

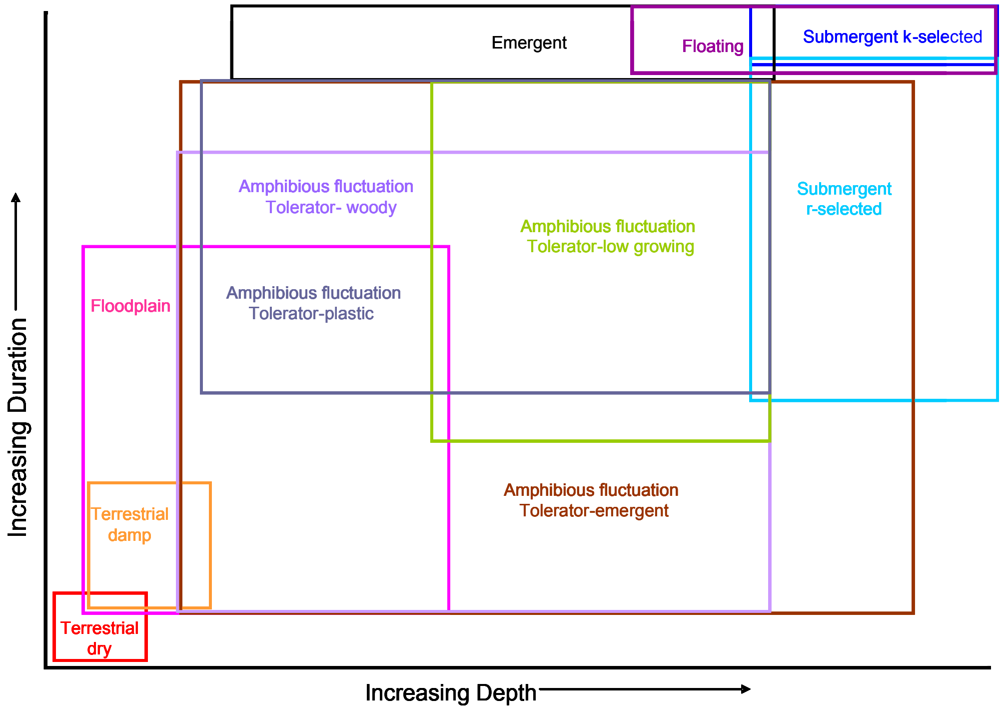

Figure 5: Plant functional groups in relation to depth and duration of inundation (taken from Nicol *et al.* (2010))

Life history strategies: Life history strategies are the traits that affect the amount of investment the plant makes in its life history stages such as reproduction, growth and survival (Begon *et al.* 1990). These life history strategies can be used to predict the likely succession of plants following a disturbance (Noble and Slatyer 1980). Changing the water regime of a wetland from being permanently to intermittently inundated may be considered a disturbance. When water levels drop for the first time, the sediments that become exposed may be bare with little or no aquatic plants. Colonisation can only occur if the seeds or vegetative propagules are present or disperse into the new habitat and that new habitat has suitable physico-chemical conditions. Unlike the vast majority of animals that are able to move at multiple life stages, for most plants their recruitment life stage (e.g. seed or vegetative propagule) is the only time they can occupy new habitats. The exceptions are floating plants that can move with wind and water at any time.

Colonisation of bare sediment patches (large and small) may occur via:

- germination of viable seed from an existing seed bank (e.g. *Rumex* sp., *Persicaria* sp., *Typha domingensis*)
- hydrocory – germination of seed brought in by water (e.g. river red gums), wind (e.g. *Typha domingensis*) or animals (e.g. *Ruppia* spp.)
- establishment of vegetative parts of plants that have moved into the wetlands with water or animals (e.g. stems of *Myriophyllum* spp.).

These life cycle strategies differ markedly between different types of plants and thus the composition and location of wetland vegetation communities will be driven by a complex interaction of abiotic and biotic factors. Noble and Slatyer (1980) determined that there are certain 'vital attributes' of plants, such as their dispersive mechanisms, that can be used to predict sequences following a disturbance (Table 2).

Table 2: Vital attributes of selected wetland plant taxa. Sources of information contained include from Casanova (2011), Nicol et al. (2010) and Roberts and Marston (2011).

Taxon	Dispersal	Recruitment	Persistence	Competition
Eucalyptus camaldulensis and *E. largiflorens*	Water Aerial seed bank in canopy	Damp soil for seed germination	Seed held on trees, Long-lived woody adults	Low – abundant seedlings, fast growing roots and shoots
Muhlenbeckia florulenta	Poor seed bank, hydrochory	Only after flood	Long –lived adults, short-lived seed bank	Low – forms dense stands
Samphires (*Sarcocornia* spp. *Suaeda australis* and *Tecticornia* spp.)	Seed bank Vegetative through lignotuber	Damp soil for seed germination	Seed bank, highly desiccation tolerant.	Low – long-lived and highly desiccation tolerant
Phragmites australis	Wind Vegetative growth	Waterlogged soil	Desiccation tolerant (underground rhizomes, thatching to protect young shoots)	Low – long-lived seed banks and underground storage organs, good disperser
Rumex sp., *Perscaria* sp.	Seed bank	Damp soil for seed germination	Seed bank	Medium-rapid growth but outcompeted by *Typha* and *Phragmites*
Ludwigia sp., *Hydrilla* sp., *Potamogeton crispus*	Seed bank Vegetative fragments, Hydrochory Animals (external)	Submergence	Seed bank	Low-occupies habitats that exclude most competitors
Potamogeton tricarinatus	Seed bank Vegetative fragments Hydrochory Animals (external)	Submergence	Seed bank	Low-occupies habitats that exclude most competitors
Myriophyllum spp.	Germinate underwater Vegetative growth Hydrochory Animals (external)	Submergence or saturated soil	Seed bank	Low-occupies habitats that exclude most competitors
Azolla spp., *Lemna* spp.	Seed bank Vegetative fragments Hydrochory Animals (external)	Submergence or saturated soil	Seed bank	Low-occupies habitats that exclude most competitors
Ruppia spp	Seed bank Vegetative fragments Animals (e.g. waterfowl) Hydrochory	Submergence	Seed bank and turions (small bulbs present on *R. tuberosa* and *R. polycarpa*)	Highly salt tolerant and germinates rapidly when sediment is inundated. Poor competitor in fresh environments

Drivers of aquatic plant communities in different wetland types

Permanent wetlands

According to the stressor models of Souter (2009) the major drivers that affect vegetation in permanent lakes and swamps are groundwater, soil type and River Murray inputs. These drive water quality (dissolved oxygen, turbidity) and surface water regime (water level, discharge) stressor levels that, in turn, may cause changes in macrophyte, phytoplankton and riparian vegetation communities.

Saline swamps

This is also the case for saline swamps in the wet and dry phases and local climate is considered an additional driver for saline swamps in the dry phase (Souter 2009). However, dry soil quality (salinity, soil moisture) is the most important potential stressor acting with surface water regime (water level, discharge) to potentially change the vegetation community and amphibious species seed bank.

Temporary wetlands

In the wet phase of temporary wetlands, groundwater, soil type, inputs and local climate are drivers of macrophytes and riparian vegetation, while water quality and surface water regime are potential stressors (Souter 2009). In the dry phase of temporary wetlands fire is a direct driver of the vegetation community and amphibious species seed bank and soil quality is a known stressor along with surface water regime.

Permanent watercourses

In permanent watercourse reaches surface water regime (season, discharge and water level) and hydraulics (flow velocity) are major stressors on macrophytes and riparian vegetation (Souter 2009). River Murray water and groundwater will affect surface water quality, particularly salinity that may be a significant stressor. Geomorphology may also affect the flora through in-channel complexity and erosion.

Ephemeral watercourses

The seasonal watercourse reach has similar controls but with different relative importance compared with the permanent watercourse reach. Season and frequency become the most important hydrological factors, which in turn determine the relative spatial and temporal occurrence of wet and dry phases for the vegetation communities. The hydrological factors, frequency and duration and their interactions with water quality and connectivity, strongly affect the flora in ephemeral watercourse reaches. It should be noted that although Souter (2009) does not list groundwater and soil condition as controls for ephemeral watercourses, there may be situations where groundwater and soil condition are significant controls.

Souter (2009) also describes a state-and-transition model for floodplain and temporary wetland understory vegetation comprising a wet and a dry phase. This suggests that surface water regime (season, frequency, duration and water level) and species interactions are stressors in the wet phase and that the most important water quality parameters are turbidity and salinity. Competition between amphibious plants and herbivory also shape the vegetation community in the wet phase but weeds are not included as a threat.

By contrast, in the dry phase flooding frequency is the most important hydrological factor because of its effects on amphibious and flood responder seed-bank viability (longer inter-flood periods experienced under regulated river conditions may lead to less viable seed banks). Evaporation and evapoconcentration can dry out the soil profile and favour terrestrial species, whilst the salinity regime can select for halophytic flora. Groundwater influx or rise can increase soil moisture availability favouring amphibious plants, but it can also be highly saline, in which case, halophytes will again be selected for. Competition between understorey plants and with invasive weeds (e.g. *Xanthium* spp.) effects vegetation during the dry phase as will shading by overstorey plants, invertebrate herbivory and grazing by vertebrates (native, pest and stock animals). The composition and viability of the seed bank will also strongly affect the resultant plant community.

Conceptualised vegetation responses to water level manipulation

The water regime required for the different plant functional groups, life histories and wetland types can be used to explain the vegetation communities present in a wetland. In a static, permanent River Murray wetland with high turbidity, plant zonation is highly constrained and relatively distinct vegetation bands are present (Figure 6). Wetlands have little or no aquatic vegetation except around the wetland margins and possibly in areas of shallow water. Woody plants have a simple age structure and there is a high proportion of open water to vegetated aquatic habitat. Furthermore, there is a high abundance of phytoplankton.

Increasing the diversity and range of aquatic plants by manipulating the water regime is a primary management objective for most of the wetland management plans in the Lower River Murray. If there is a transition from a permanent water regime to an intermittent one, the locations and times within which the water requirements of a given plant are met are likely to shift. This may cause a disturbance, which results in the re-distribution of the vegetation community.

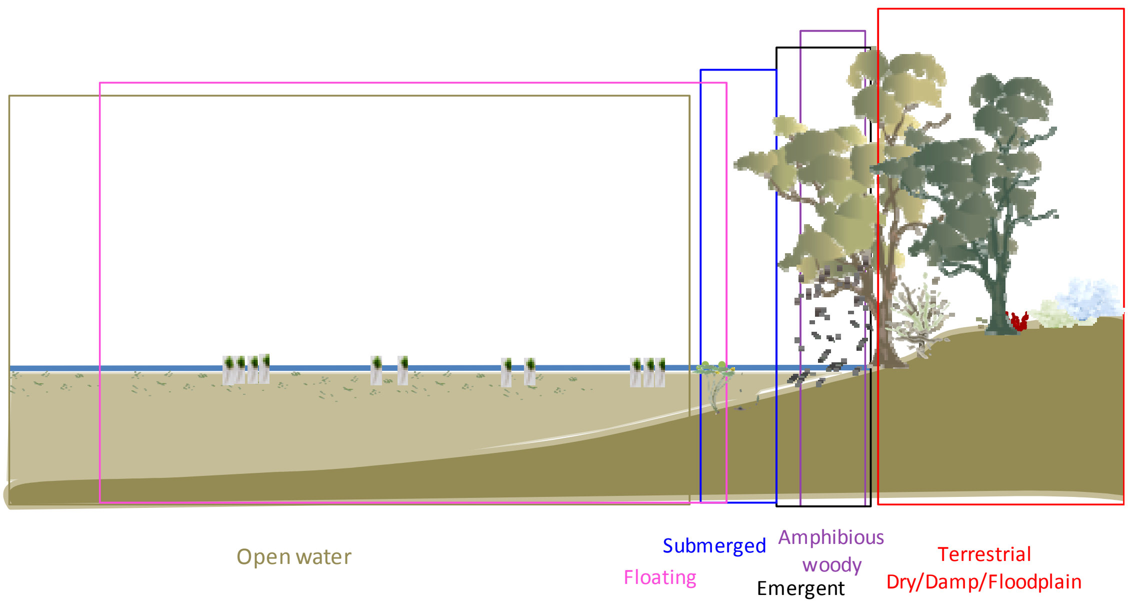

Figure 6: Typical zonation pattern of a permanent wetland with a static water regime and high turbidity

As a wetland dries, riparian plants and sediment will become exposed and the soil moisture profile will start to change as the saturated soil zone moves down the elevation gradient. A range of floodplain, dry, damp and amphibious plants will germinate on the exposed sediments and the increased availability of light may encourage clonal expansion of emergent plants if soil moisture is favourable (e.g. *Phragmites australis* and *Typha domingensis*) (Figure 7). These germinants and new shoots are likely to become established, provided that soil moisture is sufficient to support growth and no other significant adverse conditions exist (e.g. pH is near-neutral, nutrients are available, grazing pressure is low). The drawdown will act as a disturbance that may lead to increased habitat heterogeneity and patchiness in the vegetation community. The relatively distinct boundaries between the plant functional groups seen in the static water regime are likely to become less well defined and the plant groups more interconnected.

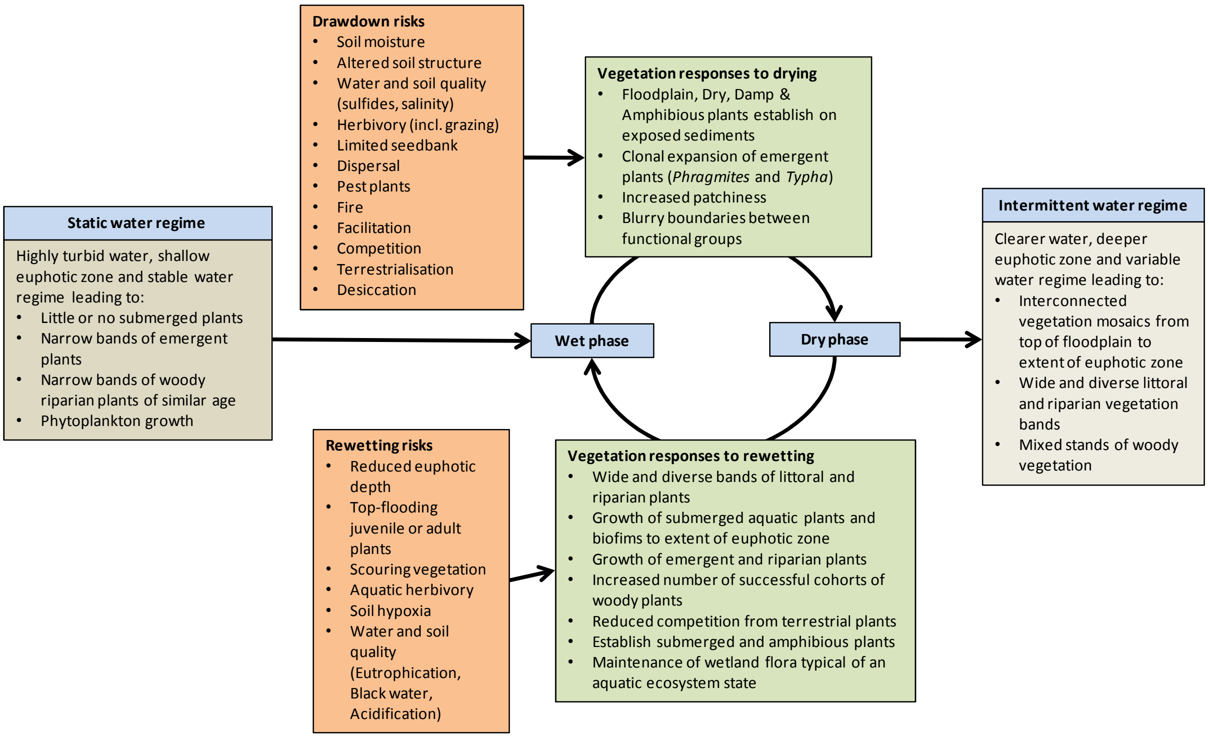

Figure 7: Conceptual model of vegetation dynamics during wet and dry phase during the transition from static to intermittent water regimes

On the floodplain the vegetation composition can exist in one of four states as determined by the period between inundations. Areas of floodplain that have been inundated within the last 12 months or the pool-level littoral zone will have high soil moisture and will support vegetation communities dominated by amphibious and flood dependent taxa (State 1), as shown in the floodplain community dynamics conceptual model developed by Nicol *et al.* (2010; Figure 8). If the floodplain is not inundated the flood dependent and amphibious plants will be replaced by drought-tolerant terrestrial species (State 2). When an area of floodplain is dominated by terrestrial taxa and inundated, the terrestrial species will be replaced by amphibious and flood dependent taxa. However if the floodplain is not inundated and accumulates salt from groundwater influx, the terrestrial species will be replaced by salt tolerant species (State 3), unless soil salinity increases to beyond the tolerances of those species, which will lead to bare soil (State 4).

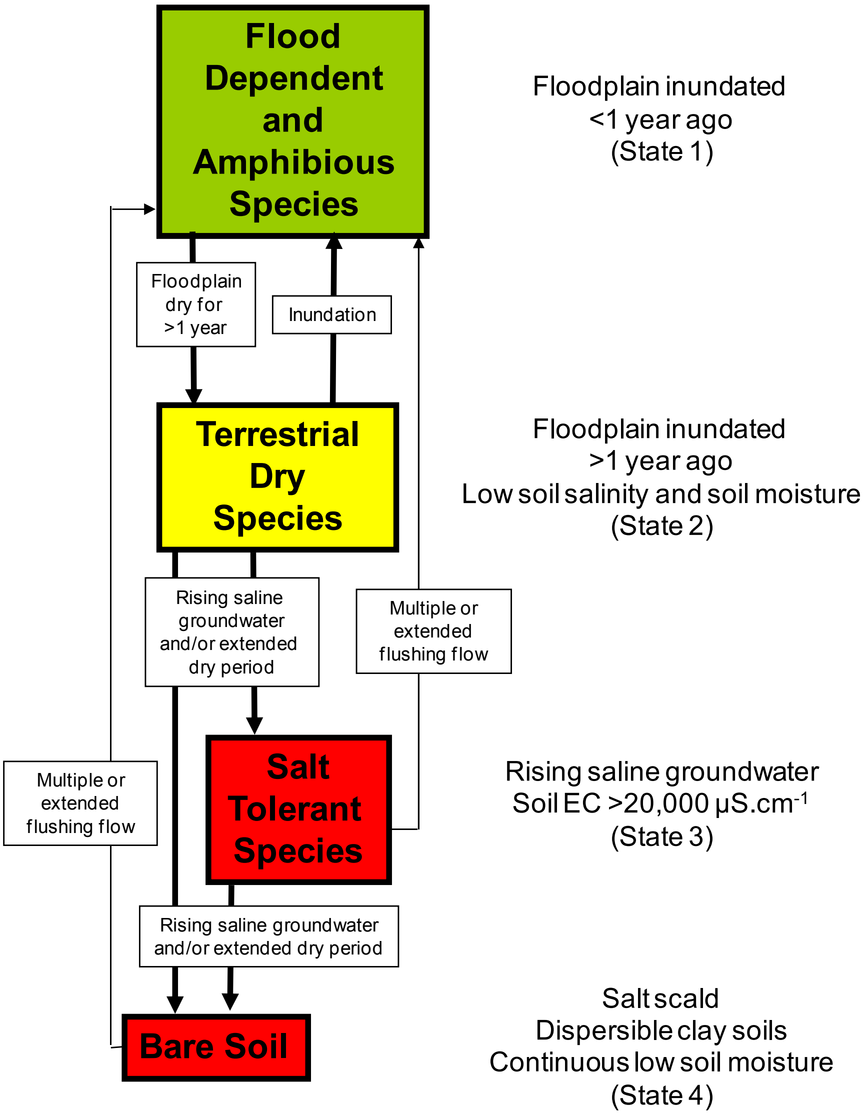

Figure 8: Conceptual model of floodplain vegetation community dynamics showing alternate states based on the functional groups present and the major factors that drive transitions in state (reproduced from Nicol *et al.*, 2010a)

Major risks to the expected vegetation response to water level manipulation (Figure 8) include:

- The drawdown phase may reduce the soil moisture available to plants. Drying can induce water stress in emergent and woody plants if their roots are unable to access soil moisture. If the drying phase is too long for a given plant it may become desiccated and die before it has a chance to complete its life cycle and replenish the seed bank.
- Water quality may also decline during drawdown as evaporation leads to concentration of compounds in the water and the optimal concentrations or tolerances of some plants may be exceeded, leading to sub-optimal growth and the

loss or exclusion of some species (see Gehrig and Nicol (2010) for salinity tolerances). Drying may also increase soil salinity through increased influx of saline groundwater (Salinity and groundwater interactions section) and may result in acidification due to the exposure of acid sulfate soils (Biogeochemical cycles section). Furthermore, extended dry phases may also alter sediment biogeochemical cycles, which may alter the availability of pollutants and nutrients that in turn may affect plant growth or community composition.

- Limited seed bank diversity and dispersal can be significant risks to achieving the expected vegetation responses. Different plants have different modes of dispersal and recruitment which will influence their capacity to colonise a new habitat. Colonisation will only occur if the seeds or propagules are present (or introduced) and conditions are suitable (habitat and resource availability).
- The riparian and floodplain zones are the most susceptible to fire. Fire across the wetland basin itself is only likely to be a risk during the dry phase if significant dry or combustible plant material is present in the wetland.
- The drawdown phase exposes the wetland to colonisation by both desirable and pest plants. Some species such as *Xanthium* spp. and *Phyla canescens* may become dominant.
- Population processes such as competition and facilitation may be important for some species, particularly those that are rare, require specific niche pre-conditions and/or have poor competitive capacity.
- If the drying phase is too short, there may be germination during the drying phase followed by top-flooding and decomposition of juvenile plants unable to grow quickly enough to match rising water levels or keep their photosynthetic tissue in the euphotic zone. For some rare species this lost reproductive effort may exhaust the seed bank or significantly reduce its viability.
- If the drying phase is too long, the wetland will become terrestrialised as aquatic species are replaced by terrestrial species. For example, terrestrialisation can result in a shift from a community of submerged, floating plants and emergent plants to terrestrial trees and shrubs that colonise from the wetland banks (Kvet *et al.* 2002).
- Upon re-inundation vegetation may be drowned out by rising water levels (top flooded). Different plants and life stages will have different susceptibilities to top flooding, with the adult phases and fluctuation responders being the least susceptible and small woody germinants and fluctuation intolerant species being the most susceptible. In the case of emergent plants, long-term submersion of photosynthetic tissue may lead to die-off from inability to supply oxygen to underground organs in hypoxic soil, as well as smothering of photosynthetic tissue by epiphytes.
- The soil and water quality changes observed upon rewetting will be highly dependent on the source water, rates of refill, dilution capacity and bio-physico-chemical properties of the wetland (Soil and water quality responses to water level manipulation section). The major water-quality risks are eutrophication from increased nutrients (source water and soil releases), acidification (from mobilisation of oxidation by-products from exposed acid sulfate soils) and low dissolved oxygen (blackwater) from the rapid decomposition of flooded organic matter (dead fauna, submerged vegetative tissue).

Likely plant succession under sequential wetting and drying cycles

Starting point: A static, turbid, permanent wetland with very little or no submerged vegetation (Figure 9). Emergent and amphibious plants form a ring around the edge of the permanent water line. Plant diversity is low, with plants that prefer static conditions favoured. The age structure of woody riparian and floodplain plants is very simple, with a probable lack of juveniles (decreasing presence of juveniles with increasing distance from the wetland edge).

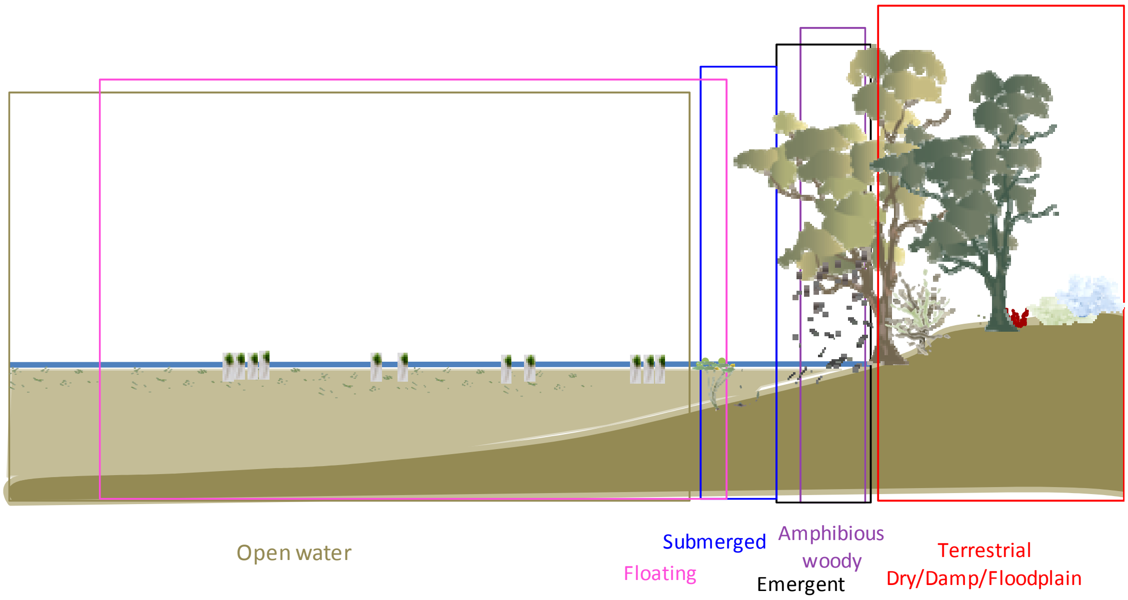

Figure 9: Typical zonation of vegetation in a permanent, static and turbid River Murray wetland

The first drawdown: When a permanent, static and turbid wetland is first dried, the sediments are likely to be relatively bare, at least in patches, particularly if the wetland is dried quickly (within a matter of weeks). This effect will be more pronounced in turbid wetlands that are greater than 0.5 m deep due to a constrained light climate resulting in low submerged plant growth.

Commensurate with the extent of the drying, the proportion of wetland basin available for terrestrial species will greatly increase upon drawdown (Figure 10). Conversely the proportion of aquatic habitat will be greatly reduced and obligate aquatic organisms will only be supported in any remaining pool or pools within the wetland if drying is incomplete. Dry, damp and terrestrial plants will germinate according to the viability and diversity of the seed bank. Moderators of the vegetation response from seed and propagule banks will be soil moisture and quality. *Persicaria lapathifolium*, *Ludwigia peploides*, *Rumex* spp., *Cyperus gymnocaulos* and a range of *Juncus* spp. are likely to germinate on the damp exposed soils on drawdown and will persist into the refill. The composition of this community of amphibious responders, their position of the elevation gradient and abundance will depend on the soil moisture regime, soil quality and the diversity and viability of available seed or propagules.

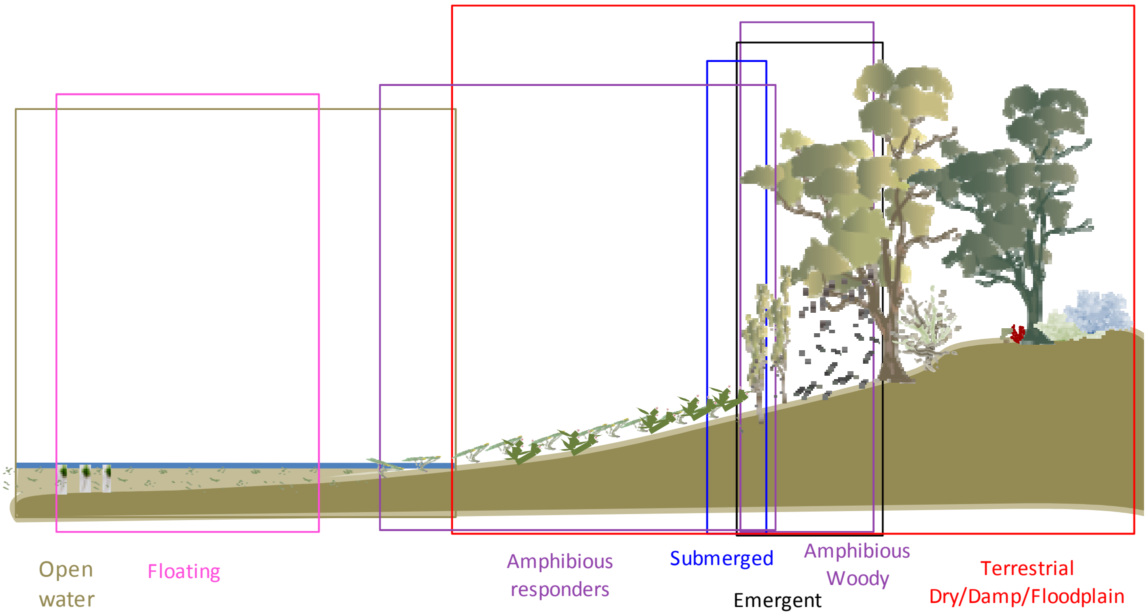

Figure 10: Likely vegetation responses when a permanent wetland is drawndown for the first time

There are over 100 species of Terrestrial Damp, Terrestrial Dry and Floodplain plants (including a range of terrestrial weeds) that may germinate on the exposed bed (see Gehrig and Nicol 2010; Gehrig *et al.* 2011; Gehrig *et al.* 2012; Marsland and Nicol 2008a, 2008b; 2009a; 2009b; Nicol 2004, 2007, 2010; Nicol *et al.* 2006; 2010a; 2010b; Weedon *et al.* 2006; Zampatti *et al.* 2011). The emergent plants, *Phragmites australis* and *Typha* spp. may colonise down the elevation gradient depending on soil moisture regime, with *Typha* spp. occupying the niches with highest soil moisture. *E. camaldulensis* are highly likely to germinate *en masse* in patches or bands around the wetland edge with the seed most likely coming from the canopies of the mature trees on the banks. If *E. camaldulensis* seedling taproot-growth can track the optimal soil moisture band as it drops through the soil profile (see Roberts and Marston (2011) for review), they are likely to become established trees, if they are not inundated on subsequent fill cycles. Terrestrial weeds (e.g. *Heliotrope* spp., *Xanthium* spp.) are also likely to colonise the bare exposed mud during the drawdown phase.

Early refilling (first 1–3 years): Upon refilling, the terrestrial habitat will retract, whilst the aquatic habitat will expand. Terrestrial plants that established on the exposed sediments will be inundated and most likely die, adding organic matter to the wetland. If they are inundated before they set seed, they will most likely decline in relative proportion within the seed bank.

The inflowing River Murray water may contain whole plants, vegetative propagules and seeds, thus acting as a dispersal agent. Floating plants will potentially have a competitive advantage because they will be among the first plants to enter the wetlands with River Murray water and they will be delivered as whole functioning plants (Figure 11). Fish screens may limit their passive movement into the wetland. The likelihood of dominance by floating plants will also depend on their cover in the remaining water in the wetland at the beginning of the rewetting cycle. If they are forming dense floating mats in the wetland and are not being dispersed by wind, then they have greater chance of establishing as the dominant plant and suppressing submerged plants. However, it is most likely that the combined wind and water turbulence will disperse the floating plants so dense mats will not cover large portions of the wetland.

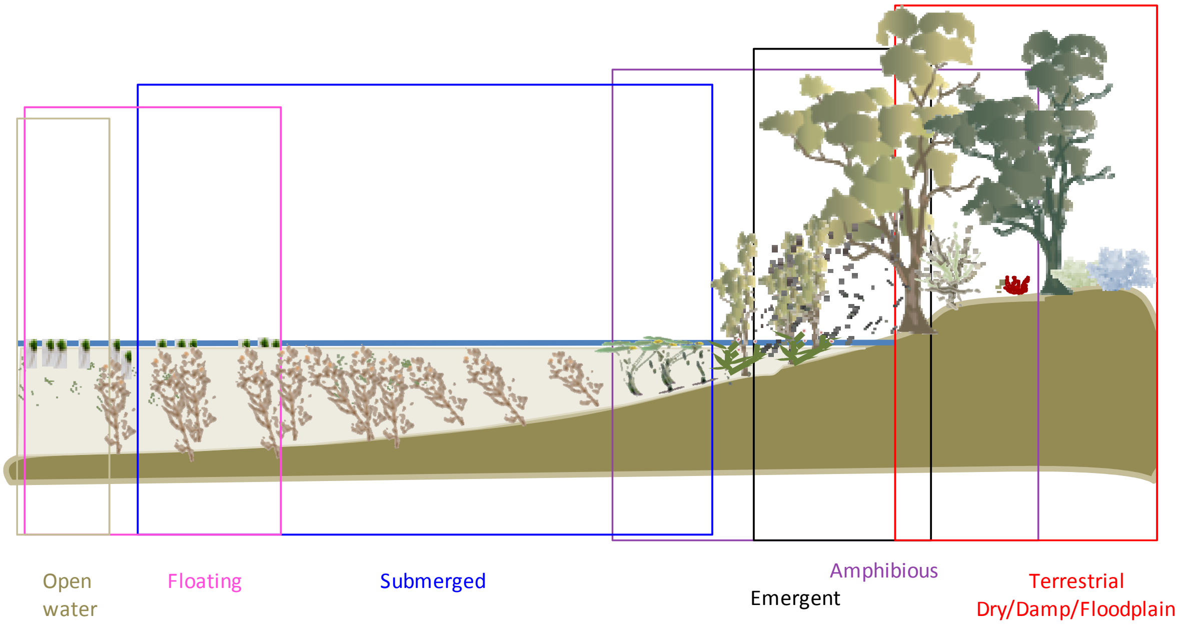

Figure 11: Likely vegetation responses to refilling following a drawdown in the first 1-3 years of introducing an intermittent water regime

Submerged plants such as *Myriophyllum* spp. will likely germinate *en masse* when the dried wetland is refilled, with dense beds forming across the wetland to a depth of approximately 0.75 m. Based on observations between 2006 and 2010–11 below Blanchetown, if no further drawdown occurs the distribution of *Myriophyllum* spp. beds is likely to decline significantly after approximately 18 months (J. Nicol, South Australian Research and Development Institute, pers. comm.).

Amphibious plants that germinated during the drawdown will survive the refilling in areas where they can match their growth to the water level rise. Their longer-term survival will depend on future water level variations. For example, *E. camaldulensis* seedlings may grow to match the incoming water, but over time they will need adequate soil moisture in drawdown phases and regular periods of soil and root aeration during the wet phases to grow and mature.

Subsequent drawdowns (2–5 years): In subsequent drawdowns it is expected that the amphibious plant band will become wider and that the proportion of Terrestrial plants that germinate on the exposed wetland basin will decrease. The proportion of Floodplain species will increase compared to the first drawdown, even though the width of the band with a terrestrial water regime will be relatively wider (Figure 12). *E. camaldulensis* seedlings that germinated on the first drawdown will grow and more will germinate if habitat is available. Submerged plants will persist as underground storage organs in the areas that are exposed provided that soil moisture remains high enough to keep them moist. Submerged plants will also persist as live plants in the remaining pool(s) of water and some species (e.g. *Myriophyllum* spp.) will add fresh seed to the seed bank. It is likely that the diversity of the Emergent plant community will have increased and will now contain species that prefer variable water regimes (e.g. *Scheonoplectus validus*, *Triglochin procerum, Juncus* spp.). By this stage the water column should be less turbid as the positive feedback loops of sediment consolidation and aquatic plant growth establish (Turbidity section).

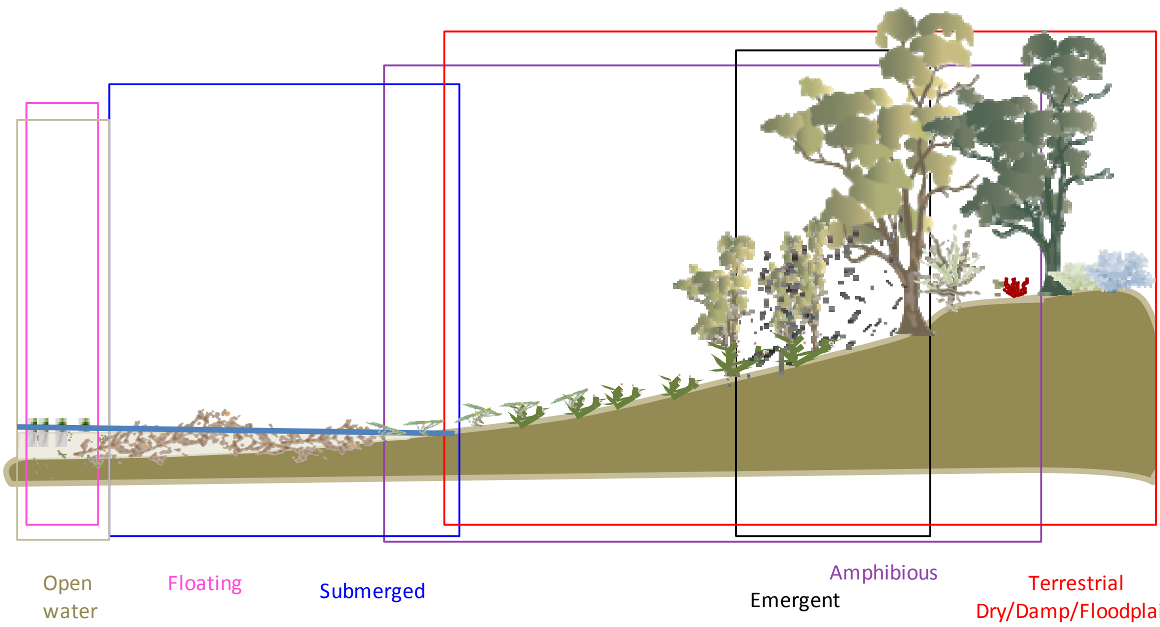

Figure 12: Likely vegetation responses to successive drawdowns in the first 2–5 years of introducing an intermittent water regime

Subsequent refills (2–5 years): In subsequent refills it is expected that the submerged plants will become denser and more diverse, although *Myriophyllum* spp. are likely to dominate in the early years. It should be noted that only 10 submerged plant species are known to occur in the region so it is difficult to predict the likely increases in diversity. The wetting phase will help to drive the relative decrease in Terrestrial taxa and the relative increase in Floodplain and Amphibious plants in the parts of the wetland basin that undergo the greatest water level variation (Figure 13). Emergent plants are likely to increase their cover, growing down the gradient. Based on observations during the low flow period in 2008 (J. Nicol, South Australian Research and Development Institute, pers. comm.), as the turbidity of the water column decreases, the diversity of the submerged plant community will increase with species such as *Potamogeton crispus* and *Hydrilla verticillata* becoming more dominant. However, there were observations of *Elodea canadensis* (a weed of national significance) recruiting during the period of low turbidity during the drought (J. Nicol, South Australian Research and Development Institute, pers. comm.).

Overall, the diversity and structure of the littoral and riparian zones will increase over time with the bands of different functional groups intermingling down the elevation gradient. This overlap of the boundaries between different functional groups will be greatest where the water levels are varied the most within the tolerance bands of the different vegetation types.

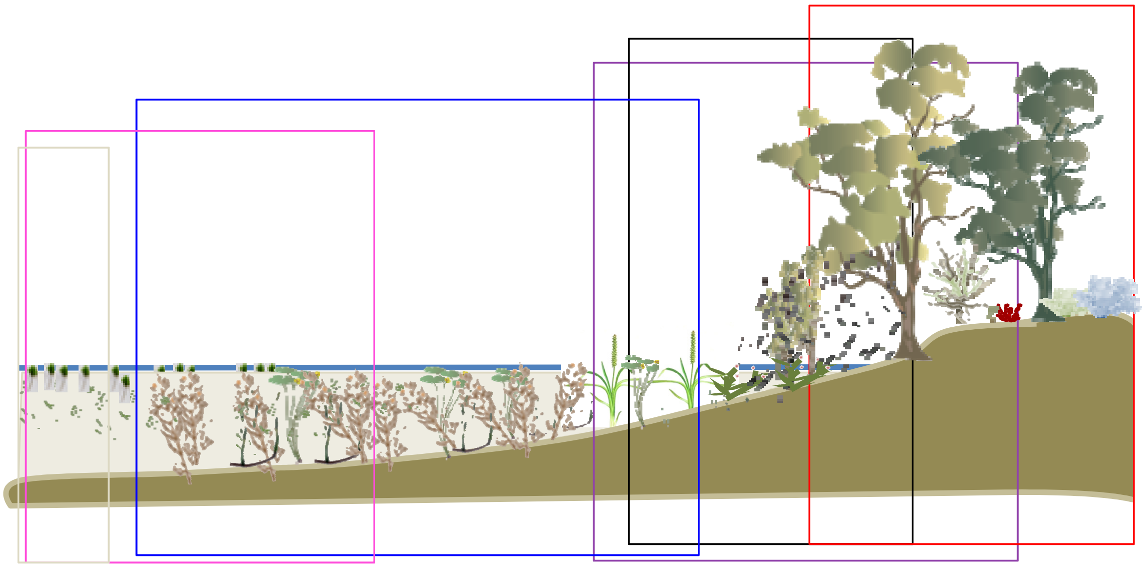

Figure 13: Likely vegetation responses to successive refills in the first 2–5 years of introducing an intermittent water regime

Vegetation community after at least 5 years of water level manipulation: Once an intermittent water regime has been implemented for at least five years, it is expected that the wetland vegetation communities will be more diverse, more abundant and that the boundaries between the groups will be less distinct. The aim is for an interconnected mosaic of different plant functional groups from the terrestrial edge of the high floodwater mark to the extent of the euphotic zone on the wetland side (Figure 14). It is difficult to determine under which conditions maximum diversity of vegetation species would be supported. However, it is important to recognise that maximum diversity in the vegetation community is not an 'end-point' or indeed desirable for every wetland at all times. Across the landscape there will be wetlands with managed water levels that have different vegetation assemblages, which together support a range of habitats and diverse fauna. It is also important to recognise that communities are dynamic and will shift over time in response to changing climatic and water regime conditions. This confers ecological resilience and will enhance the capacity of Lower River Murray wetlands to adapt to changing conditions.

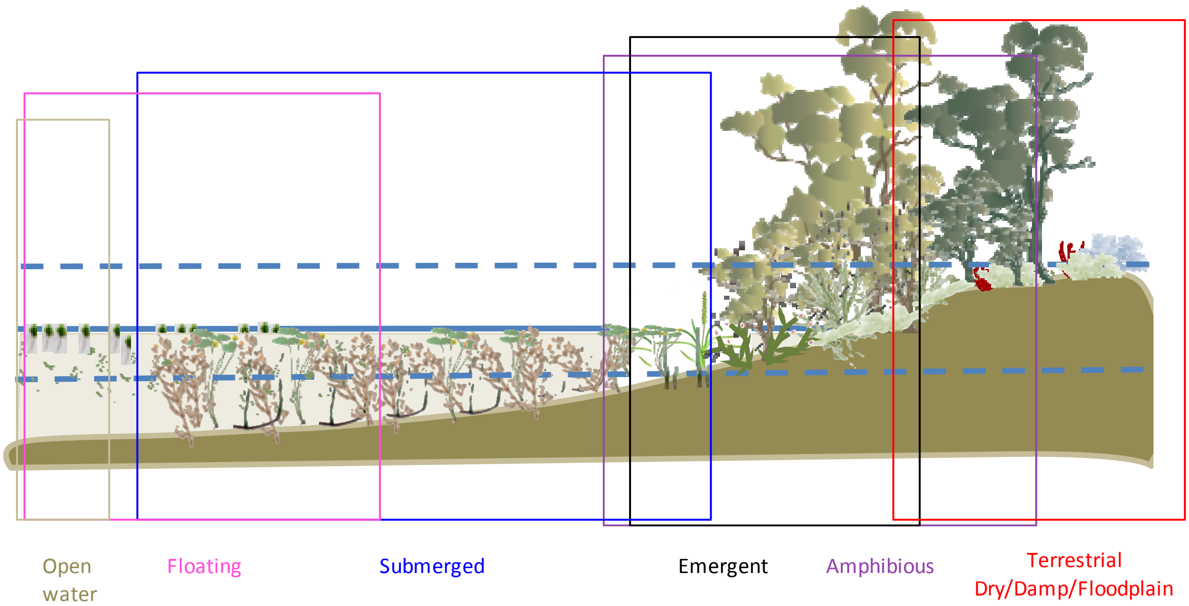

Figure 14: Likely vegetation community structure after more than five years of applying a variable and intermittent water regime to a previously permanently inundated and turbid River Murray wetland. The upper dashed line shows the high water level obtained under flood conditions or by weir pool manipulation. The lower dashed line shows the average drawdown level, although water levels may drop lower than that during some drawdowns.

Impacts of weir pool raising (1 in 10 years): The water level variations described above will not have changed the condition or composition of the Floodplain, Dry and Damp species that occur on elevations higher than can be influenced by wetland regulatory structures. The only way that these higher floodplain areas will be inundated is if there are high River Murray flows that fill the wetlands above pool level (floods), or if the pool level in a given reach is raised by increasing the height of the downstream weir (weir pool raising).

Raising of the water level will inundate floodplain soils higher up the elevation gradient (Figure 15). Adult *Muhlenbeckia florulenta* (lignum) are expected to improve in health as are mature *E. camaldulensis* and *E. largiflorens,* both of which are also likely to flower and set seed which will be retained in the canopy seed bank. Woody plants that are low to the ground (e.g. samphires) may become inundated and will become stressed if the water level covers the majority of their photosynthetic tissue for long enough to induce die off and decomposition. This may be beneficial for controlling terrestrial weed species that are not adapted to inundation. Water level raising is unlikely to significantly alter the width of the different functional group bands.

Once the floodwaters recede, a wide range of Floodplain, Dry and Damp species are expected to germinate (Figure 16). Provided the soil moisture regime is suitable, these germinants should grow and mature, to provide additional cohorts and improve population age structure. Overall, there will be a tendency towards diverse floodplain vegetation communities with a mixed age structures. Floodplain vegetation communities have the highest possible diversity of all the functional groups with approximately 100 species recorded in the region (J. Nicol, South Australian Research and Development Institute, pers. comm.). Whilst it is difficult to predict what species will occur during or after a given water level raising, it is likely that the optimal period of return will be every two to three years. Less frequent inundation will see the floodplain increasingly terrestrialised and on-going decline in Floodplain and Amphibious Woody species.

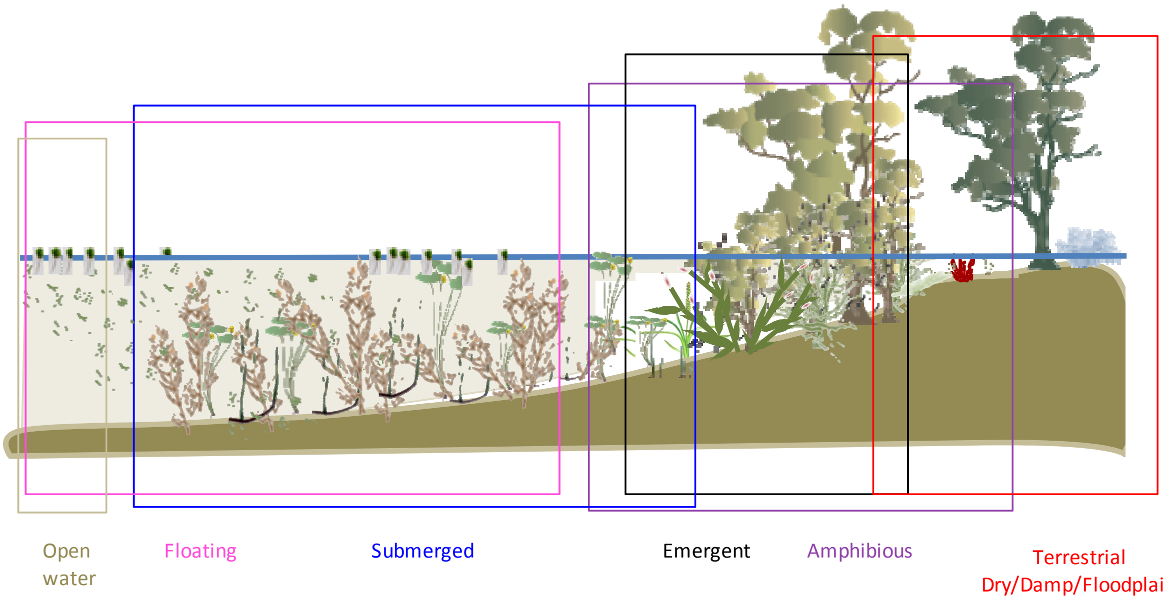

Figure 15: Likely vegetation responses to flooding or raising of wetland water levels via weir pool manipulation

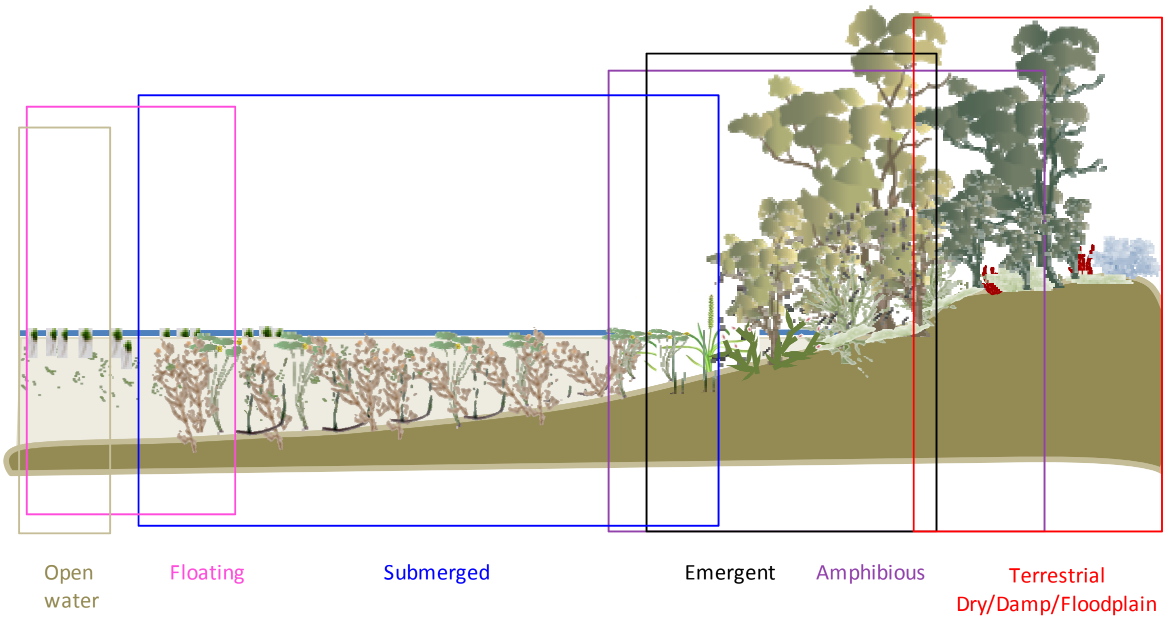

Figure 16: Likely vegetation responses to the recession following flooding or raising of wetland water levels via weir pool manipulation

Consumer responses to water level manipulation

Background

Consumers are organisms that depend on other organisms (e.g. living or dead plants, algae, animals, fungi and bacteria) for their energy because they are unable to manufacture their own food from inorganic materials (Begon *et al.* 1990). They can be grouped based on their diet (grazers or predators: Boulton and Brock (1999)) or their prey specificity (generalists or specialists: Begon *et al.* (1990)). In a taxonomic sense consumers can be further classified based on whether they are vertebrates or invertebrates (Boulton and Brock 1999).

Invertebrate grazers occur in the littoral and open-water zones feeding on periphyton (biota attached to submerged surfaces), phytoplankton and plants. Open-water invertebrate grazers, such as water boatmen, feed on particulate plant matter as well as detritus and smaller invertebrates. Zooplankton are by far the most important open-water grazers (Boulton and Brock 1999). They graze on bacteria and phytoplankton and in turn become prey for invertebrate and vertebrate consumers. Vertebrate grazers include tadpoles, fish and water-birds.

Predators can occur at the surface of open-water (e.g. water bugs and beetles) or in the littoral zone (e.g. dragonfly larvae and beetles). The architectural complexity of the littoral zone can shelter diverse prey species and thus attracts diverse predator communities. Many predators eat each other, resulting in food chains that are typically topped by large vertebrate predators such as large-bodied fish, Water-rats, freshwater turtles or water-birds. Water-birds are the most mobile of these predators being able to fly from wetland to wetland seeking food, shelter and nesting habitats. Water-birds such as ibis may consume terrestrial prey (e.g. grasshoppers, caterpillars) and terrestrial predators may consume prey from the wetland environment, particularly if the wetland is dry or if wetland animals enter the floodplain zone (e.g. water-birds, freshwater turtles). Thus trophic interactions are not confined to the wetland and may provide a link between aquatic and terrestrial habitats through which energy, carbon and nutrients are transferred.

In permanent wetlands, the habitats available to a given consumer will be relatively well defined by water regime and the distinct and narrow bands of vegetation. Opportunities for colonising new habitats will be limited, competition may be high, productivity low and food chains well-defined and stable.

In intermittently flooded wetlands, resource availability can be much greater. For example, as a wetland fills there will be abundant resources and little competition for those consumers who can disperse and establish in the filling wetland. Early colonisers will benefit from the boom in primary production as aquatic habitat increases with rising water levels and nutrients are released into the water column (Biogeochemical cycles section). Depending on the period of inundation, intermittently flooded wetlands may favour consumers with short life cycles or short aquatic life stages and high mobility. However, exact consumer assemblages, trophic cascades and biotic sequences will depend on wetting and drying history, particularly the extent and duration of previous wetting and drying cycles (e.g. Puckridge *et al.* 1999; Souter 1996; Souter *et al.* 2000).

The following section conceptualises how frogs, fish and water-birds might respond to transitioning a permanent, static wetland to an intermittently flooded wetland. Typically these are secondary consumers feeding on primary consumers such as zooplankton and macroinvertebrates. Many consumer groups are not examined in detail. This is not a reflection of their importance to wetland ecology but rather reflects the need to align monitoring effort with management objectives with limited resources. Management objectives are largely focused on frogs, fish and water-birds (as well as vegetation).

Frogs

Background

Frogs are amphibious consumers that are typically associated with water bodies (lakes to small depressions). Eleven frog species are present along the River Murray corridor within South Australia (Table 3). Whilst frogs are opportunistic carnivores, tadpoles predominantly feed on vegetation and sediment, but will also opportunistically prey on insects or dead tadpoles (Anstis 2007). Although there are apparently no inter-species differences in dietary preferences, significant differences exists between species in habitat associations, preferred water regime and capacity to tolerate habitat drying (Table 3). Frogs and tadpoles strongly associate with wetland vegetation, because it provides habitat resources such as food, sites for egg laying for some species and cover from predators such as fish, snakes and water-birds. Male frogs will typically call whilst afloat on the open water, floating on algal or plant mats, from vegetation near the edge of the wetland or from depressions or ditches near wetlands (see Anstis (2007) for review and species specificity where known).

Table 3: Vital attributes of frogs (adapted from Anstis (2007); Gonzalez *et al.* (2011); Turner *et al.* (2011); SAMDB NRMB unpubl. data; Mason and Turner, Department of Environment, Water and Natural Resources, pers. comm.). Qualitatively 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 aquatic vegetation and low salinity	Recently inundated vegetation; Males call: spring to autumn from vegetation; Metamorphosis: summer to autumn, 2.5 to 15 months; Tadpoles present in Nov., absent from Feb. onwards.
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 mild winters from vegetation (after heavy rains); Metamorphosis: 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 Nov.
Common froglet (*Crinia signifera*)	Dominant below Lock 1; habitat generalists; prefer abundant aquatic vegetation or submerged terrestrial vegetation; Desiccation	Breed opportunistically; dispersal poorly understood; Males call most of the year; Tadpole maturity: 6 weeks to 3 months or more,

Species	Habitat associations	Recruitment needs
	avoidance poorly understood.	absent by Nov.; Tadpole salinity tolerance up to 9, 360 µS/cm.
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 µS/cm.
Eastern banjo frog (*Limnodynastes 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: 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; aestiviate 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; aestiviate 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 after rain. Tadpoles: often overwinter after autumn breeding and metamorphose in spring and early summer.
Brown Toadlet/Bibron's Toadlet (*Pseudophryne bibronii*)	Not dependant on river;	Males call: February to June, particularly after heavy rain; usually next near wetlands or creeks; tadpole maturation: 120–180 days, metamorphosing late winter to summer.

Note: there are no known pest frog species in South Australia

Different frog species have unique responses to environmental cues and their tadpoles have different inundation requirements from adult frogs of the same species. For example, male *Litoria raniformis* (southern bell frog) 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 to 15 months (Wassens 2005). By contrast, the highly opportunistic *Crinia* spp. will rapidly disperse following rain at any time of year to make use of temporary habitats that may only be wet for six weeks, which is sufficient for tadpoles to mature (Anstis 2007). It is important to note that although males may call, successful recruitment will not necessarily follow.

As well as the requirement for inundated habitats for the period of tadpole maturation, Newman (1998) found that temperature and food availability interacted in determining the rate of metamorphosis and the size of the tadpole at metamorphosis. Newman (1998) concluded that if adults breed too late in the wet phase, environmental conditions may not be suitable for tadpoles to mature and become adult frogs, which are relatively tolerant of sub-optimal conditions. This is consistent with observations at Clayton Bay in 2009. Southern bell frog males were found in wetlands with salinities greater than 10,000 µS/cm, however, no tadpoles were captured. This suggested that although the males were trying to breed, tadpoles were not successfully recruiting at those salinities. Indeed, southern bell frog tadpoles were only detected in low salinity water (400-600 µS/cm; SAMDB NRMB unpubl. data). In comparison, other species may have greater environmental tolerance or adaptability such as *Limnodynastes* sp. and *Crinia* sp. tadpoles, which were found in Lakes Alexandrina and Albert at salinities up to 9,360 µS/cm (SAMDB NRMB unpubl. data).

Overall, poor water quality is expected to lead to adverse impacts on frogs. For example, hypoxia has been shown to lead to deformities and death of embryos and hatching at earlier stages of development of some frog species. This early hatching is likely to have a negative impact on growth, ability to avoid predation and reproductive success at adult stages (Mills and Barnhart 1999; Seymour *et al.* 2000).

Of the eleven species present, southern bell frog is the only nationally threatened species (Vulnerable listing under *EPBC Act* 1999) although several other species have State or regional listings. 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 southern bell frogs suitable habitat for other native frog species is also likely to be provided. Observations of southern bell frog habitat requirements include:

- calling males were lowest during periods of low river flow
- mostly observed after natural or artificial flood events
- highly mobile, moving up to two km between wetlands, especially during floods
- dependent on regular flooding to promote recruitment
- reside near permanent refuge sites without flooding
- sensitive to drying and drought (mass mortality can occur when refugia dry)
- strongly associated with complex wetland vegetation and appear to prefer lignum, river red gum or black box, diverse emergent plants and herbs (e.g. *Eleocharis* sp., *Ludwigia peploides*), floating and submerged plants (e.g. *Myriophyllum* spp., *Marsilea* spp., *Azolla* spp. and *Lemna* spp.) and inundated grasses (Gonzalez *et al.* 2011; Turner *et al.* 2011; Wassens 2011).

In addition, statistical analysis of monitoring data indicated southern bell frogs were more likely to be found in wetlands with conductivity of less than < 471.5 µS/cm; with emergent vegetation cover of between 5–50% and in recently drained, previously permanent wetlands (Souter 2011). Sites that did not contain southern bell frogs were predominantly wetlands with medially sloping banks, dead river red gum overstorey, very dense reeds (e.g. *Typha* sp. and *Phragmites australis*) or no mid-storey and had salt tolerant plant species in the understorey (Schultz 2006 as cited in Turner *et al.* 2011).

Based on this information by providing complex vegetation across most of the elevation gradient and regular flooding (Table 3), suitable habitat will be provided for southern bell frogs and other native species. Sites in the Lower River Murray support this, as all sites where southern bell frogs were calling also had a high diversity of other frogs (K. Mason, Department of Environment, Water and Natural Resources, pers. comm. 2012).

Conceptualised frog responses to water level manipulation

The regulation of the River Murray, in particular static wetland water regimes and reduced flooding, have seen many frogs decline in distribution and abundance over the last few decades (e.g. southern bell frogs, Long-thumbed frog; Gonzalez *et al*. 2011). This is most likely due to dependence on flooding for dispersal and successful recruitment and preferences for well-vegetated temporary or ephemeral wetlands, as opposed to static and turbid permanent wetlands (Table 3). Baseline surveys support this preference for variable water regime with a greater number of sites with southern bell frogs in the Riverland (Lock 3 to the SA border) where a large number of managed wetlands occur (SKM 2006a). The impact of a static water regime on different frogs is likely to be highly dependent on the topography, depth and water quality of a given wetland.

Drawdown events: When a wetland dries, tadpoles that have obligate aquatic life stages are at risk (Figure 17) if water does not persist until the tadpoles have fully metamorphosed into frogs. If this is the case, tadpole may be trapped and desiccated or heavily preyed upon as the aquatic habitat reduces. Tadpoles are the most susceptible life stage to the changes in water quality that occur with wetland drying (e.g. increased salinity and temperature). However, adults may also be directly and indirectly affected, particularly if they are unable to find suitable food, vegetation cover or retreat to permanent refugia (e.g. dense riparian vegetation along the River Murray channel: Tyler 1994). There is a trade-off for tadpoles under an intermittent water regime, between capitalising on a potentially unlimited food supply to ensure a large size at metamorphosis and the risk of staying in an ephemeral pool, desiccating and being eaten (Rose 2005).

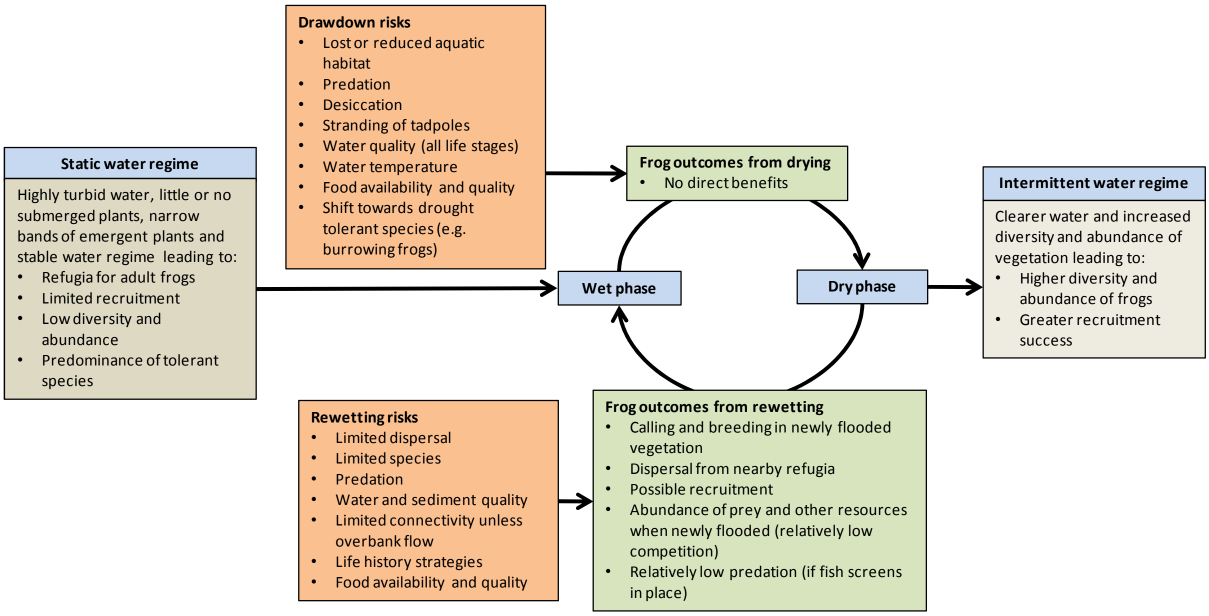

Figure 17: Conceptual model of frog responses to wetting and drying phases during the transition from static to intermittent water regimes

Different frogs are likely to respond differently to drying. 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 comparison, 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 500 m to 1 km during drawdown will most likely increase resilience and local persistence. Conversely, eastern banjo frogs emerge to forage during heavy rains, which may extend the length of time individuals can persist between floods (Wassens 2011). These frogs may be sensitive to desiccation and their distribution may be restricted by soil type. Therefore, if the dry phase is too long - and rain insufficient - then eastern banjo frogs will need to retreat to permanent refugia and rely on dispersal to re-colonise the wetland upon re-wetting.

Peron's tree frog and the spotted grass frog are probably the species most resilient to extended drying, being found at isolated wetlands when they re-wet after being dry for extended periods (see Gonzalez *et al.* (2011) for review). Although there are no direct benefits of drying to these frogs there may be indirect benefits, such as increased diversity and abundance of aquatic vegetation and extirpation of predatory fish.

Refilling events: When a wetland is re-wet, within days male frogs are likely to begin calling from newly inundated vegetation (K. Mason, Department of Environment, Water and Natural Resources, pers. comm. 2012). These frogs will have either dispersed from nearly permanent refugia, other temporary wetlands or emerged from moist microhabitats within the wetland. Abundant food and habitat availability will encourage breeding, which can lead to successful recruitment if the wetland supports diverse and abundant vegetation.

The risks on re-wetting are adverse water and sediment quality early in the refill phase; seasonality of re-wetting not supporting life cycle completion; and predation (Figure 17). An additional risk is the lack of population connectivity, particularly for species such as murray valley froglet that benefit from, or depend on, wider inundation generated by overbank flow (weir pool raising or flooding) for successful breeding. It is important to note that although re-wetting may induce calling by male frogs at almost any time of year, recruitment will be most successful if the wetland is filled in later winter and remains inundated until late summer/early autumn.

Wetlands that have a prolific aquatic vegetation response to refilling will provide particularly good habitat for frogs because of the abundance of food and the limited predation and competition by fish. 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, Department of Environment, Water and Natural Resources, pers. comm. 2012). Similarly, weir pool raising or flooding that inundates depressions behind main wetland basins may also support abundant frog populations (e.g. Sweeney's Lagoon in 2006, K. Mason, Department of Environment, Water and Natural Resources, pers. comm.).

Fish

Background

Fish are obligate aquatic organisms: they have no life stages that can persist without water and thus they are dependent on the quality, quantity and functional connectivity (hydraulic and population) of inundated habitats for their survival.

Whilst the classification of fish into different functional groups continues to be debated, fish can be split into six categories, based on body size, habitat use, response to hydrology and conservation status (adapted from Bice and Ye (2009)):

- Common large-bodied native fish
    - Native species with an average adult body length >150 mm that complete their life cycle in freshwater (e.g. Murray cod, Golden perch, Bony herring)
- Rare large-bodied native fish
    - Native species with an average adult body length >150 mm. Are considered rare (possessing a state or national conservation significance) in the Lower River Murray (including the lower Lakes) and complete their life cycle in freshwater (e.g. freshwater catfish, silver perch and river blackfish)
- Common small-bodied native fish
    - Native species with an average adult body length <150 mm. Are considered common in the Lower River Murray (including the Lower Lakes) and complete their life cycle in freshwater (e.g. murray rainbow fish, un-specked hardyhead, carp gudgeons);
- Rare small-bodied native fish
    - Native species with an average adult body length <150 mm. Are considered rare (possessing a state or national conservation significance) in the Lower River Murray (including the Lower Lakes) and complete their life cycle in freshwater (e.g. southern purple-spotted gudgeon, murray hardyhead)

- Diadromous fish
    - Both catadromous and anadromous species: they require access between freshwater and marine habitats to complete their life cycles. The highly disconnected nature of the regulated river, lakes and Murray Mouth are likely to limit their capacity for upstream migration, thus the freshwater life stages are more likely to be abundant below Lock 1 (e.g. congolli, short-headed lamprey)
- Alien freshwater fish
    - Alien species that complete their life cycle within freshwater environments that can be large bodied or small-bodied (e.g. carp, redfin perch, eastern gambusia, goldfish, weather loach).

Life-history models for most native and exotic fish species that occur in the River Murray are presented in Bice (2010). Species-specific habitat associations, diet preferences and recruitment needs have also been identified (Table 4).

Table 4: Vital attributes of selected fish species (adapted from Bice 2010; Smith *et al.* 2009 and Wedderburn *et al.* 2007)

Species	Habitat associations	Diet preferences	Recruitment needs
Large-bodied freshwater native fish			
Murray cod (*Maccullochella peelii*)	Riverine habitat with in stream cover, deep holes. Larvae usually in fast-flowing habitats	Carnivore: Adults – fish, frogs, crustaceans, microcrustaceans, water-birds, turtles Larvae - crustaceans, microcrustaceans	Spawn: spring and early summer (>15°C), may be flow-related, eggs deposited on hard surface, guarded by males; Do not need flow to initiate spawning but flow may facilitate recruitment
Golden perch (*Macquaria ambigua ambigua*)	Turbid and slow river channel, associated with structure; Wide range of habitats, also found in wetlands and floodplains	Opportunistic carnivore; Adults – fish, insects, crustaceans Larvae, juveniles - microcrustaceans	Spawn: spring and summer (>20°C), Spawning in rivers is flow-related, semi-buoyant and non-adhesive eggs
Bony herring (*Nematalosa erebi*)	Common and widespread; variety of habitats including open water	Algal detritivore – detritus, algae, microcrustaceans	Spawn: late spring and summer (>20°C); shallow bays
Rare large-bodied freshwater native fish			
Freshwater catfish (*Tandanus tandanus*)	Slow-flowing turbid streams and wetland habitats, Often associated with fringing vegetation, demersal species	Opportunistic carnivore: Adults – fish, insects, crustaceans, Larvae - microcrustaceans	Spawn: spring and early summer (20-24°C), Non-adhesive eggs spawned onto nest of pebbles, gravel or sand
Silver perch (*Bidyanus bidyanus*)	Turbid and slow-flowing river channel, large lakes, also found in wetlands	Omnivorous; Adults – aquatic plants, snails, insects,	Spawn: spring and summer (>23°C), increases during floods, Semi-buoyant eggs

Species	Habitat associations	Diet preferences	Recruitment needs
		crustaceans; Larvae - microcrustaceans	
Common small-bodied freshwater native fish			
Carp gudgeon complex (*Hypseleotris* spp.)	Larvae found in various habitats, Adults in wetlands usually associated with aquatic vegetation	Generalist omnivore; Adults - detritus, insects, fish larvae, microcrustaceans, algae. Larvae - rotifers	Spawn: spring and summer (>20°C), Adhesive eggs on structure, male guards eggs
Murray rainbowfish (*Melanotaenia fluviatilis*)	Slow-flowing rivers, streams and wetlands. Often associated with aquatic vegetation.	Opportunistic carnivore; Adults – insects, microcrustaceans Larvae – plankton, microcrustaceans	Spawn: spring and summer (>20°C), Adhesive eggs laid on submerged vegetation
Unspecked hardyhead (*Craterocephalus stercusmuscarum fulvus*)	Littoral habitats of slow-flowing rivers and wetlands, Often associated with aquatic vegetation.	Carnivore; Adults – aquatic insects and zooplankton, microcrustaceans Larvae - rotifers	Spawn: spring and summer (>24°C), Adhesive eggs attach to submerged vegetation
Australian smelt (*Retropinna semoni*)	Various slow-flowing or still habitats – channel, wetlands, pelagic	Opportunistic carnivore; Adults – zooplankton and insects, Larvae - rotifers	Spawn: late winter and spring (>11°C), Adhesive eggs on structure
Rare small-bodied native fish			
Murray hardyhead (*Craterocephalus fluviatilis*)	Off-channel shallow (<0.5 m) wetlands, sheltered margins, Often associated with aquatic vegetation (e.g. *Ruppia* spp.); tolerates relatively high salinities (3-20 ppt or approximately 5-32 mS/cm)	Omnivores; Adults – zooplankton, microcrustaceans, insects, algae; Larvae - rotifers =	Spawn: spring and summer, Adhesive eggs attach to submerged vegetation
Southern purple-spotted gudgeon (*Mogurnda adspersa*)	Demersal species, slow-flowing areas of streams and wetlands, Often associated with abundant cover	Benthic carnivore Adults – insects, microcrustaceans, fish Larvae - rotifers	Spawn: summer (20-30°C), Adhesive eggs on structure, Male guards eggs
Diadromous fish			
Congolli (*Pseudophritis urvillii*)	Larvae: estuarine/marine, Juveniles: migrate from estuarine to fresh. Adults: off-channel wetlands, main channel, sandy and mud substrates	Benthic carnivore, Adults – fish, crustaceans, insects Larvae – Marine and estuarine zooplankton	Catadromous Spawn: autumn to spring, Possible habitat segregation of males and females that needs to be overcome
Short-headed lamprey (*Mordacia*	Ammocoetes: soft substrates in Lower River Murray (below Lock 1, perhaps further upstream),	Ammocoetes: filter feeders. Adults: parasitic on fish	Anadromous Spawn: adults migrate upstream to spawn in winter-spring in

Species	Habitat associations	Diet preferences	Recruitment needs
mordax)	Wetlands, slow-flowing water near edge, Adults: mostly marine		shallow habitats
Short-finned eel (*Anguilla australis*)	Larvae: ocean, Juveniles: Lower River Murray and Lakes, Adults: slow-flowing rivers, lakes or wetlands	Carnivore; Adults – fish, crustaceans, molluscs, insects	Catadromous Spawn: near New Caledonia in the Coral Sea
Alien freshwater fish			
Carp (*Cyprinus carpio*)	Larvae and juveniles: wetlands and floodplain. Adults: Variable, slow-flowing rivers, wetlands, vegetation and open water	Omnivore – plant material, detritus, insects, molluscs, microcrustaceans, crustaceans	Spawn: spring and summer (>16-17°C), Adhesive eggs on structure, Can spawn for up to 6 months (spring to autumn)
Eastern gambusia (*Gambusia holbrooki*)	Slow-flowing and still habitats, Often associated with aquatic vegetation	Carnivore – insects, zooplankton	Spawn: spring and summer, bear live young in slow-flowing waters
Redfin perch (*Perca fluviatilis*)	Slow-flowing habitats, Often associated with aquatic vegetation	Carnivore – crustaceans, fish	Spawn: spring (>12°C), Eggs in ribbons amongst vegetation
Goldfish (*Carassius auratus*)	Slow-flowing rivers and wetlands, Often associated with aquatic vegetation	Omnivore – plant material, detritus	Spawn: summer (>17°C), adhesive eggs on structure

The homogenisation of River Murray wetlands connected at pool level that has occurred due to regulation (Pressey 1986; Walker 2006) is reflected in the fish communities, which are very similar between the river channel and the wetlands and are dominated by generalist species (Smith *et al.* 2009). For example, small-bodied native fish such as carp gudgeons and un-specked hardyhead that would be expected to utilise wetland habitats in unregulated systems have been regularly found in fish surveys of the main river channel, except in times of variable flow (C. Bice and Q. Ye, South Australian Research and Development Institute, pers. comm.). Downstream of Blanchetown (Lock and Weir 1) the river typically has a more variable hydrology and is better connected to estuarine and stream habitats. Thus, it provides habitat heterogeneity and contains the most diverse native fish assemblages in the South Australian section of the River Murray (Smith *et al.* 2009). In comparison, above Blanchetown the river contains more larger, stable, permanent, relatively unproductive wetlands, with fish assemblages containing mostly 'generalist' native fish species (Smith *et al.* 2009).

Many wetland specialist species with conservation listing (e.g. murray hardyhead, purple-spotted gudgeon, southern pygmy perch and yarra pygmy perch) prefer wetlands and their demise may be due to lack of intermittent wetland habitat, the loss of lotic riverine conditions, or both (Lintermans 2007). Also the homogenisation of wetlands habitats and reduced productivity caused by the maintenance of stable water regimes may have also played a part in their demise.

The different responses of fish species to regulation is highlighted by the comparision between the distribution of murray hardyhead and un-specked hardyhead. Murray hardyhead have a wide but patchy distribution, whereas un-specked hardyhead have a similar range but are typically more abundant. Furthermore, the two species rarely cohabit. Wedderburn *et al.* (2007) suggest that this separation is due to salinity, with murray hardyhead more salt tolerant (Table 4). The habitats for murray hardyhead (well-defined by Wedderburn *et al.* (2007)) are:

- several hundred river kilometres upstream of the Murray Mouth, murray hardyhead are highly abundant in low diversity assemblages occurring in habitat that is shallow (<0.5 m), saline (3–20 ppt, approximately 5–32 mS/cm), completely or partially disconnected from the river channel, with little woody debris and a few macrophyte species (predominantly *Ruppia* spp.).
- in the lower reaches of the river (mostly below Blanchetown), murray hardyhead occur in low numbers within diverse fish assemblages dominated by diadromous and estuarine species in wetlands that are shallow (<0.5 m), slightly saline (ca. 1 ppt or 1600 µs/cm µS/cm), partially or fully connected to the channel and dominated by the submerged plant, *Myriophyllum* spp..

These observations suggest that murray hardyhead dominate in saline wetlands in the upper reaches of the River Murray in South Australia due to competitive advantage over less saline tolerant species rather than a preference for saline water *per se*.

Carp are the most abundant alien fish, occurring in self-sustaining populations in all parts of the Murray-Darling Basin (Smith *et al*. 2009). Their adverse impacts on wetlands and native fish species are well recognised and include: competing for resources (Cadwallader 1978); increasing turbidity, nitrogen concentrations and phytoplankton biomass by 'mumbling' (King 1995); and damaging shallow-rooted, soft-leaved submerged plants (Fletcher *et al.* 1985; King 1995; Roberts *et al.* 1995). These adverse impacts are most consequential in lentic, well-vegetated, warm, shallow habitats such as wetlands (Gehrke *et al.* 1995; Koehn *et al.* 2000) where carp may congregate to breed (Jones and Stuart 2008; Stuart and Jones 2006). These impacts can transition wetlands from macrophyte-dominated clear water states to phytoplankton-dominated turbid water states (Matsuzaki *et al.* 2009). Many native fish species depend on macrophyte cover for habitat (Table 4).

Carp are 'first-on last-off' a wetland as it refills or dries (Jones and Stuart 2008). Mature carp may make large-scale spawning migrations and move from deep water into shallow wetlands to spawn, behaviours which may be exploited to control their numbers. Juvenile carp have more restricted home ranges than adult carp (see Smith (2005) for review). These characteristics are being further researched to find improved ways of managing wetlands to control carp. One way of controlling carp numbers is to completely dry the wetland to kill all the trapped fish. It may be possible to reduce losses of native fish when this occurs by exploiting differences in the timing of carp and native species to leave wetlands during drying events. The feasibility of implementing such an operational regime, and the resulting impact on fish communities if it can be achieved, are yet to be tested.

Native fish movement between the main river channel and wetlands is generally bi-directional and somewhat haphazard. It is thought that fish use wetlands to exploit food resources (e.g. zooplankton, macroinvertebrates, smaller fish), avoid predation and as nursery grounds. Most native fish are highly opportunistic generalists. They may respond to wetting and drying cycle cues in wetlands (for example, emigrating during a drying phase) and changes in River Murray flow, moving between wetlands and the main channel as resources and habitats change. In the upper River Murray (between Lake Mulwala and

Hume Dam) fish moved into wetlands on a rising flow in the river and back to the main channel when flows in the river were decreasing (Lyon *et al.* 2010). In contrast during the low flow period in the Lower River Murray in 2006, no apparent directional movements between main channel and connected wetlands were observed, potentially due to low-flow conditions (Conallin *et al.* 2011). Regardless of preferences for lateral movement, population and hydraulic connectivity will be lost for fish when a wetland regulating structure is closed to induce and maintain a drawdown phase.

Flow and/or water regime, habitat, physico-chemical parameters and connectivity are important abiotic drivers that affect life history stages and population dynamics of fish and subsequently shape fish assemblage structure (Bice 2010; Ye *et al*. 2009). Of these, flow/water regime is the main driver as it influences the physico-chemical conditions, habitat and connectivity (hydraulic and population). Thus, the water regime in a wetland can affect fish directly or indirectly via changes in wetland habitats, water quality and productivity levels.

Conceptualised fish responses to water level manipulation

In wetlands with static water regimes and high turbidity, there may be diverse and/or abundant native fish communities, but they are likely to be dominated by a predictable suite of generalists (Smith *et al.* 2009). Conditions in permanent and static wetlands are likely to be sub-optimal for recruitment and for those species that depend more strongly on associations with submerged and amphibious vegetation (Table 4). The vigour of native fish in static, turbid wetlands may limited by the simplicity of the habitat architecture (floral and physical), relatively low food resources after long periods of constant inundation (Boulton and Brock 1999), poor recruitment success and high rates of competition with exotic fish such as carp and eastern gambusia (Figure 18).

Management of flow and water regimes in river and wetland habitats through opening and closing of regulating structures and weir pool manipulation may assist in reversing this trend by creating habitat diversity. The drying of wetlands below Blanchetown between 2006 and 2010 followed by the refilling and reconnection from 2010 onwards created large variations in these typically permanent wetlands. Observations of fish distributions suggest that shifts in fish assemblage structure towards that expected in a flowing river may be occurring (C. Bice and Q. Ye, South Australian Research and Development Institute, pers. comm. 2012).

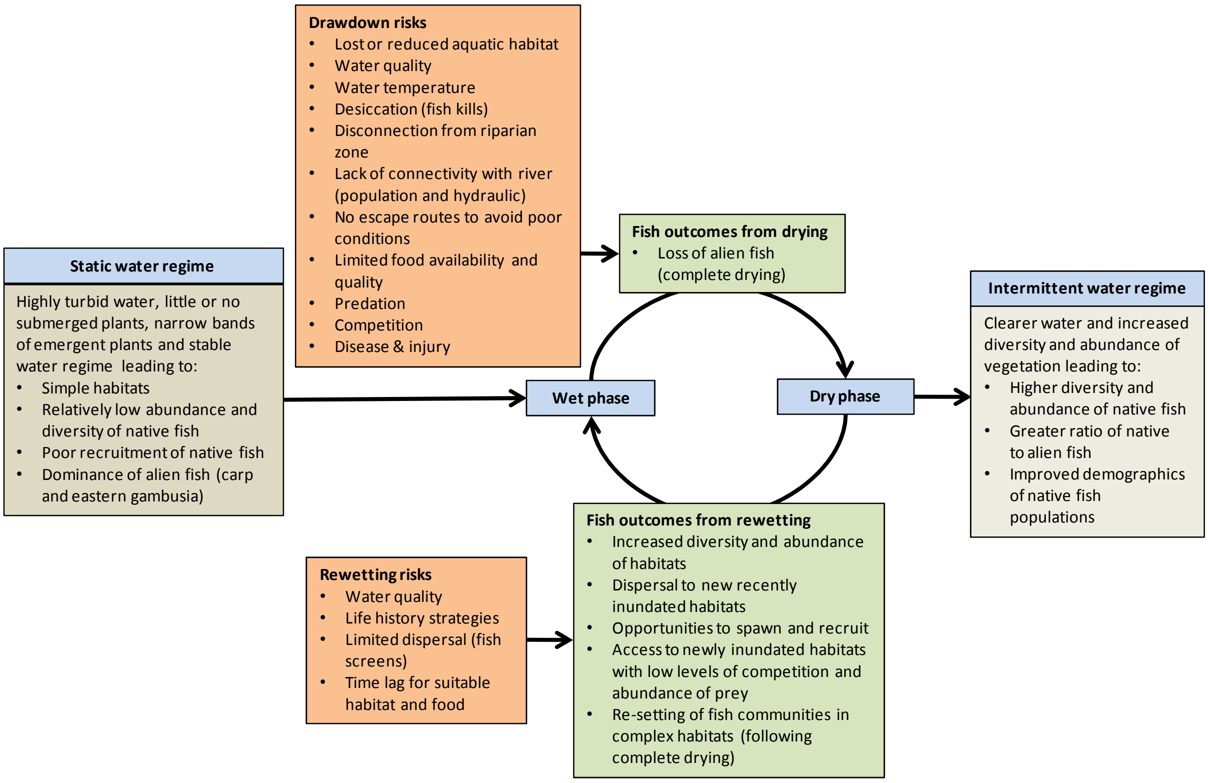

Figure 18: Conceptual model of fish responses to wetting and drying phases during the transition from static to intermittent water regimes

Drying events: Once a wetland regulating-structure is closed to initiate drawdown, no fish will be able to exit the wetland (native or alien). As a wetland dries the amount of aquatic habitat will decline, increasing the pressure on the retained fish communities. If the wetland is partially dried, then the fish contained within it will be able to take refuge in the remaining aquatic habitat and may be able to hide from predators in and around habitat structures, particularly in wetlands that have dense aquatic vegetation that remains inundated. However, in many wetlands the fish will be trapped in a water body that is disconnected from the vegetated littoral zone and thus will have relatively little cover from predators.

The decreasing water volume in a drying wetland is likely to result in changes in water quality. Water quality parameters such as salinity can affect fish directly at sub-lethal and lethal levels (McNeil and Closs 2007) or indirectly through changing and/or reducing availability of habitat and food resources (James *et al.* 2003). Temperatures are expected to rise, particularly if the drawdown occurs in the warmer months. Higher temperatures lead to higher respiration rates and lower solubility of oxygen in the water, which may lead to low dissolved oxygen levels and fish kills. Fish tolerances and behavioural responses to low dissolved oxygen differ between species. Most fish species (even juveniles) can tolerate levels of dissolved oxygen as low, or possibly lower than 3 mg/L for short periods (McNiel and Closs 2007). Some fish such as carp gudgeons, goldfish, carp and eastern gambusia can survive periods of hypoxia by employing air-surface-respiration (McNiel and Closs 2007).

Similarly, reduced water volume will lead to increased levels of salinity, nutrients and other pollutants, resulting in sub-lethal effects or possible mortality if concentrations exceed specific tolerance levels of fish species. Ye *et al.* (2010) found that the early life stages of fish are the most sensitive and vulnerable to increased salinities and suggest that their tolerance values should be used as management triggers. Water quality may become too poor to sustain fish populations or increase competition, predation and stress so that mortality, disease or injury may become prevalent even if significant areas of water remain.

If a wetland is completely dried, all fish contained within it will perish providing food for piscivorous birds, other predators and decomposers. Although, the loss of native fish in a complete dry may be a short-term negative outcome, there may be short- and long-term benefits for a wide range of native fish that results from a more variable water regime (e.g. reduced numbers of large carp immediately after refill, improved submerged vegetation habitat over the years). That said the complete drying of wetlands should only occur when the resident fish community composition and abundance is known and within the context of regional conservation initiatives.

Refilling events: The drying and then re-wetting of wetland habitats is likely to support successful recruitment of native fish (Cadwallader 1978). Rewetting will stimulate the growth of aquatic vegetation, which will provide structure, food for herbivores and harbour prey for different types of fish. Incoming fish, or those that survived the drawdown, will be able to exploit newly inundated habitats with low levels of competition and high abundance of food resources. The pulse in productivity that follows wetland rewetting is highly beneficial to native and alien fish by providing abundant biofilms, plankton and invertebrate prey and refugia from alien species such as redfin perch, carp and eastern gambusia (Smith *et al.* 2009). There may be time delays for some food and habitat resources to develop to a suitable stage that they may be used by fish (e.g. late succession macroinvertebrates). Variations in the timing, frequency and duration of inundation and connectivity will influence fish species abundance and dominance, such that the resultant community is less easy to predict than for permanent wetlands (Smith *et al.* 2009).

Generally speaking, fish will enter the wetland as it fills although there may be different cues for different fish and multiple entry pathways. A risk to fish communities upon reffling is water quality, which may cause sub-lethal or lethal effects. In particular, low dissolved oxygen levels (e.g. blackwater events) may occur after the first inundation, which may cause fish mortality. High nutrient levels in a refilled wetland may cause algae blooms, which could have direct or indirect impacts on fish (e.g. toxicity, hypoxia). If the wetland had large areas of exposed acid sulfate soils in the drawdown phase or other pollutants accumulated, there may be run-off of toxic substances into the water column. Conversely, the poor water quality may be detected by the fish and prevent them from moving into a wetland (Gehrke 1991).

As a management response to carp, screens have been fitted to most inlet structures on managed wetlands along the River Murray in South Australia. Their main purpose is to prevent the movement of carp into the wetland during the refill and wet phase. However, large carp have been found in wetlands with fish screens fitted to the inlets (Nichols and Gilligan 2003). This may be due to large carp entering the wetlands when the screens were in place (e.g. by jumping the structures) or juveniles passing through screens and maturing in the wetlands. Unless complete drying is implemented every 1–2 years, carp will become concentrated in wetlands that have screens on their inlets because the larger fish will not be able to escape (Hillyard *et al.* 2010).

Whilst existing fish screens exclude large carp they also exclude native and other alien large-bodied fish species: e.g. golden perch, bony herring, redfin perch and goldfish (Hillyard *et al.* 2010). Optimised carp screens have been designed to prevent the movement of carp ≥ 250 mm long (below median size of sexual maturity), while minimising impacts on native fishes commonly found traversing wetland inlets. Sexually mature large-bodied fish (e.g. golden perch) species will be unable to move through the fish screens and into the wetlands to spawn. Although their need to spawn in wetlands is largely unknown (K. Hillyard, Department of Environment, Water and Natural Resources, pers. comm.), juvenile fish have been found in wetlands (e.g. Lake Carlet; Smith and Fleer 2007).

Water-birds

Background

Water-birds (i.e. birds that rely on wetlands and the riparian zone) are generally highly mobile and opportunistic consumers that can move between wetlands (up to a global scale) as food and habitat resources change. They are indirectly affected water regime through changes in food and habitat resources. Access to suitable habitat and food resources at the appropriate time is essential for survival and recruitment. Overall, the composition of water-bird communities at a wetland will reflect the availability of food suited to those species' physical adaptations (e.g. beak length; Kingsford and Porter 1994; McDougall and Timms 2001).

If there is little environmental variability (i.e. a reach with many permanent wetlands), water-birds may settle randomly (Orians and Wittenberger 1991). They will be showing a response to broadscale environmental conditions, rather than indicating that all microhabitat components essential for successful recruitment are available at that site. Other human factors, such as use of farming machinery, watercraft or shooting, may disturb birds in areas causing them to concentrate in less disturbed areas.

Water-birds can be classified into six groups based on feeding and habitat requirements, as follows (adapted from Waanders and Kuchel (2011)):

- Water fowl
    - Rely on submerged vegetation and aquatic invertebrates (e.g. swans, ducks, geese, gallinules, coots)
- Waders and shorebirds
    - Forage by walking in shallow water (approximately 0.3 m) or over exposed shorelines while searching for prey (e.g. ibis, spoonbills, egrets, sandpipers)
- Piscivorous birds
    - Hunt for fish while flying over or sitting upon a water body (e.g. grebes, darters, cormorants, pelicans, egrets, bitterns, herons)
- Cryptic birds
    - Reliant on dense riparian vegetation for cover (e.g. rails, Australasian bittern)
- Gulls and terns
    - Hunt while flying and dip to the water surface to pick off prey (e.g. silver gulls, gull-billed and white-winged terns)
- Reed-dwelling passerines
    - Although technically not water-birds, these birds use riparian reeds beds for habitat (e.g. reed-warblers, grassbirds).

A number of other birds depend on the long-lived river red gums that occur on the edges of channels, wetlands and the floodplain, including the regent parrot, white bellied sea-eagle and whistling kite. Both the regent parrot ('Vulnerable' EPBC Act 1999) and white bellied sea-eagle ('Endangered' NPW Act 1972) are conservation listed and should benefit from a water regime that is suitable for river red gums and fish communities (for the piscivorous white bellied sea-eagle and whistling kite).

The associations in Table 5 are common habitats where birds have been observed. Although habitat and food requirements are relatively common among members of each group (Table 5), there are differences between species that will affect the responses to changes in water regime (Gonzalez *et al.* 2011). Water-birds are very mobile and use a wide variety of foraging habitats and different habitats at different life stages (Ecological Associates 2009). For example, when some adult water-birds are tending nestlings, they will only move short distances to forage and thus require species-specific nesting and foraging sites in close proximity to one or more wetlands (Rogers and Paton 2008). Others water-birds require nesting sites that are surrounded by water to provide protection from terrestrial predators (Marchant and Higgins 1990).

A wetland must have both appropriate nesting habitat and sufficient food (within or nearby) to meet the high energetic requirements of breeding birds for recruitment to be successful (Rogers and Paton 2008). As such, the water regime is a strong driver of recruitment success for water-birds. In order for breeding to be successful, the wetland must remain inundated long enough for the given species' life cycle to be completed. The length of breeding time varies between species. For example, yellow-billed spoonbill requires 61–66 days to complete incubation and fledge nestlings, whereas black swans can take up to 180 days (Waanders and Kuchel 2011).

Conceptualised water-bird responses to water level manipulation

Static, permanent wetlands may lower water-bird diversity and abundance than wetlands with intermittent water regimes (Figure 19). Under permanent inundation, low productivity and simple habitats results in low food and habitat availability for most water-birds. Many diving piscivorous birds have benefitted from river regulation due to the predominance of permanent, deep-water habitats (Marchant and Higgins 1990). However, other birds such as waders and shorebirds are excluded from foraging in deep and permanent waters that do not have shallow, productive margins (Rogers and Paton 2008). Similarly, ducks and swans can rarely breed on large, deep and turbid permanent wetlands due to lack of suitable habitat (Rogers and Paton 2008). Furthermore, the preferred foraging habitats for larger water-birds (e.g. egrets, ibis, and spoonbills) have been reduced by the reduction in floodplain inundation frequency since regulation and salinisation of wetlands (Gonzalez *et al.* 2011).

Drying events: Upon drawdown, mud flats, edges of fringing vegetation and detritus will become exposed and whilst still moist, will provide ideal feeding grounds for waders, shorebirds, crake and rails that feed on the invertebrates in these microhabitats (Waanders and Kuchel 2011; Figure 19). Some species feed in shallow encroaching or receding water, others on recently exposed sand or mud and some at the moving interface of water over the substrate (Ecological Associates 2009). Furthermore, aquatic prey such as fish will become concentrated and hunting for piscivorous birds will become more energetically favourable (i.e. less effort for same food supplies). Competition for food and habitat resources may increase, if many birds arrive to exploit the easily caught prey.

Table 5: Vital attributes of water-birds (adapted from Ecological Associates (2009); Waanders and Kuchel (2010); Gonzalez *et al.* (2011))

Species	Habitat associations	Diet preferences	Recruitment needs
Water fowl	Inundated wetlands	Most waterfowl: Submerged vegetation and associated aquatic invertebrates; Australian wood duck and shelduck: terrestrial foragers.	Aquatic invertebrates important food for egg and nestling development; Require nesting sites surrounded by water
Waders and shorebirds	Shallow margins of freshwater and brackish wetlands; exposed mudflats	Very broad within and between species. Gallinules, coots: largely herbivorous; Ducks and swans: associated with vegetation may eat fauna	Various habitats as a group; spoonbills: nesting trees surrounded by water Red-capped plover: dry clay and sand for nests
Piscivorous birds	Most utilise permanent, deep water; Grebes: can also use shallow temporary wetlands.	Mostly small-bodied and juvenile large-bodied fish; Some adult large-bodied fish	Darters and cormorants: live and dense river red gums or river coobah trees; Cormorants: prefer <10 m tall river red gums
Rallids	Many types of freshwater wetland; Rails and crakes: permanent to ephemeral, fresh to saline with dense fringing vegetation	Rails and crakes: Invertebrates; Terrestrial and palustrine rallids: omnivorous, predominately carnivorous	Breeding success linked to seasonality and duration of flooding and density of vegetated nesting sites.
Gulls and terns	Gulls: large range of habitats connected to water; Terns: vegetated and open water, brackish and saline lakes	Insects, amphibians, small fish, crustaceans	Silver gull: highly adaptive, not known to breed in RM wetlands Terns: floating nests made out of plant material (rarely observed in SAMDB)
Reed-dwelling Passerines	Dense stands of *Phragmites australis* and *Typha* spp.; Quiet and thus rarely observed	Insects	Breed during spring-summer Some migrate away during winter

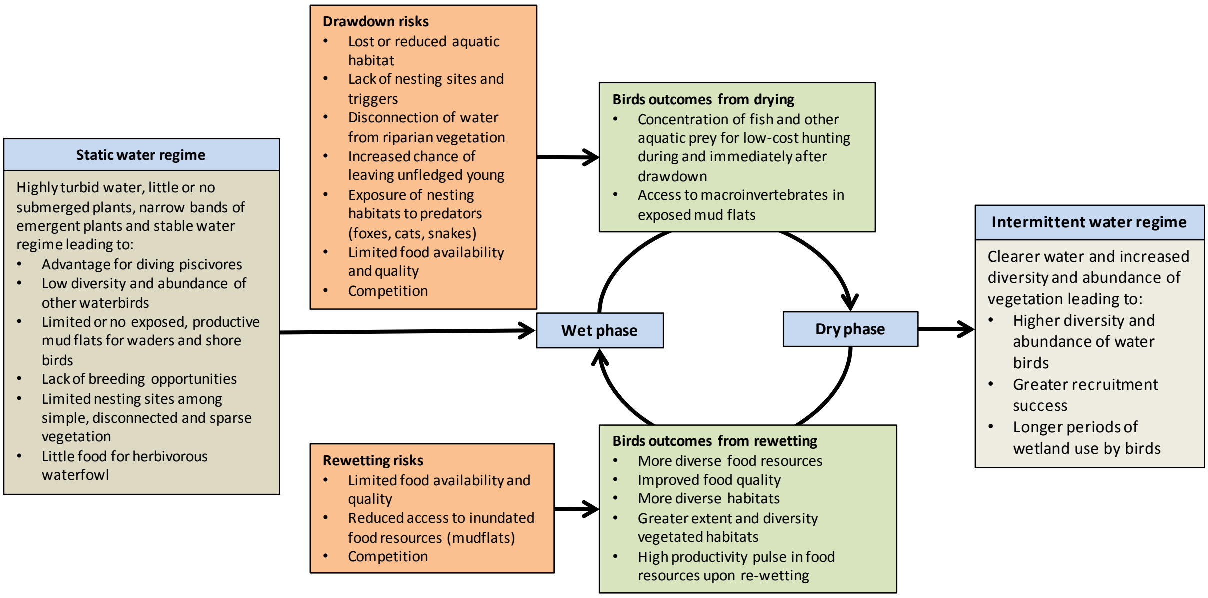

Figure 19: Conceptual model of bird responses to wetting and drying phases as a static wetland transitions to an intermittent wetland

The disconnection of fringing vegetation from the remaining water body will increase exposure of cryptic birds that like to take cover in the vegetation, which may reduce the distance they will forage and make it more difficult to detect their foraging trips.

The timing of drying is important because the nests of rallids, piscivores (e.g. grebes) and other water-birds (e.g. spoonbills and ibis on floodplain areas) may be abandoned if water levels drop or fluctuate significantly during the breeding season (Marchant and Higgins 1990); or if adults are forced to forage too far to the receded water's edge. Drying a wetland during the breeding season may also make some nests, and fledglings contained within, accessible to predators like foxes and cats.

Refilling events: upon refilling, it can be expected that wetland productivity will increase and aquatic plants, fish and invertebrates will be more diverse and abundant than in the permanent, static phase. Wetlands that wet and dry may be critical habitats for waterfowl (ducks and swans) that are unlikely to breed on permanent wetlands (Rogers and Paton 2008) and for waders and shorebirds that require exposure of mud flats and productive shallow margins. Species such as yellow-billed spoonbills are particularly responsive to inundation (floods) and will only breed if certain thresholds of overbank flow are met and water remains under nest trees for five to 10 months following flooding (Gonzalez *et al.* 2011).

The increase in aquatic vegetation diversity and abundance expected under suitable wetting and drying cycles (Vegetation response to water level manipulation section) will provide improved outcomes for many birds through increased provision of food (e.g. invertebrates) and habitat (e.g. dense and contiguous lignum, dense and large reed beds). The variability in water regime will also enhance the patchiness of different vegetation functional groups and bare mud. This will provide birds' habitats for foraging, protective and nesting habitats in close proximity. Thus it is expected that wetlands with a variety of water depths and vegetation associations will support a greater diversity and abundance of water-birds (Broome and Jarman 1983).

Overall, most water-birds will benefit from a transition from a static to an intermittent water regime. That said, it is important that not all wetlands are transitioned, nor drawn down at the same time and that a variety of wetland habitats (including monospecific stands of *Phragmites australis* for Australasian bitterns) are provided across the landscape.

Soil and water quality responses to water level manipulation

Background

Soil and water quality are major drivers of the floristic and faunal composition of wetlands. This is primarily because the growth of organisms cannot occur if their essential resources are not available and/or substances are present at concentrations beyond an organism's physiological tolerance range. Drying and rewetting wetlands and floodplains that have had extended periods of inundation or drying due to regulation will have significant implications for soil and water quality. These changes may limit the desired ecological response to water level manipulations of the RRP. Important processes that will affect soil and water quality as a result of water level manipulation are described below.

Salinity and groundwater interactions

Aquatic ecology and salinity

Salinity is a fundamental determinant of the ecological character wetlands, whether it is surface water salinity, soil salinity or groundwater salinity (Hart *et al.* 1991, 2003; Neilsen *et al.* 2003). Salinity can increase in wetlands due to source water inputs (surface or groundwater) or leaching of salts from the surrounding soil profile (Salinity and groundwater interactions section). Due to its physiological effects on biota, salinity can directly limit the range and type of biota that can survive or reproduce within a given environment (Lester *et al.* 2011). The threshold for impacts on a given taxa tends to change across their life history, with consideration of the most sensitive phases (usually juvenile) being critical for population management (James *et al.* 2003; Ye *et al.* 2010). As well as absolute salinity levels, the rates of change in water salinity may also influence the survival and vigour of wetland populations (Lester *et al.* 2008). Prior reviews have presented physiological thresholds for many aquatic biological components (Hart *et al.* 1991, 2003; Neilsen *et al.* 2003).

Increasing salinity can also affect biota indirectly, by influencing the physical (e.g. stratification) and chemical (e.g. nutrients) environment and interactions between biota. Also, the effects of changes in salinity will often occur in synergy with other stressors such as altered water regimes or increasing temperature. For example, the combined effect of salinity and waterlogging has a greater detrimental impact on growth and survival of young river red gum plants than either factor alone (James *et al.* 2003). However these indirect and synergistic effects are less well understood than direct salinity tolerances (Nielsen *et al.* 2003).

The direct and indirect effects of salinity can influence biotic population dynamics, connectivity, habitat complexity and ecological processes (Hart *et al.* 2003; Nielsen *et al.* 2003). This may lead to changes in community structure and function such as: loss of species essential for habitat or food provision, altered predation pressure and chains, limited or no successful recruitment or the depletion of propagules, such as microinvertebrate eggs and aquatic plant seeds (Nielsen *et al*. 2003). Over time there may be a shift in species composition in a given habitat in response to a change in salinity (Lester *et al.* 2011).

Lower biological diversity is often observed in wetlands with higher surface water salinity, particularly towards hypersaline concentrations. Tucker *et al.* (2003) states one of three golden rules for wetland management is to avoid salinisation. However, increases in salinity are inevitable during periods of disconnection from the river. This is likely to have been a natural feature in some wetlands prior to European settlement. Provided physiological tolerance thresholds are not exceeded, increasing salinity or variable salinities may advantage certain native species adapted to fluctuating salinity (Greenwood and McFarlane 2009; Wedderburn *et al.* 2007). It follows then that, at a community-level, the relationship between salinity and biodiversity is not a simple negative correlation (James *et al.* 2003) and a variety of salinity regimes across the landscape will be required to optimise biodiversity.

Soil sodicity is often associated with salinisation. Wallace and Rengasamy (2011) also differentiate saline and sodic soils. They state that "sodicity is a term used to describe conditions where there is a high proportion of exchangeable sodium cations relative to other common cations, primarily magnesium, potassium, calcium, and aluminium. When magnesium and calcium cations bound to clay particles in saline soils are displaced by sodium and leached from the soil profile, the sodium cations remain attached and accumulate in the soil, leading to the formation of sodic soils (Rengasamy and Marchuk 2011). A soil is classified as sodic when the level of exchangeable sodium present affects the soils structural stability. Sodicity at the surface and sub-soil layer has the capacity to affect soils via surface crusting and high soil strength. This can reduce seedling emergence and root penetration. High sodium concentrations reduce or preclude the ability of plants to restrict sodium uptake, allowing sodium to accumulate in the leaves of plants and become toxic. In addition, high sodium levels can interfere with potassium and calcium uptake. Both of these elements are essential to plants. Calcium assists with selective uptake, allowing plants to take up potassium and exclude sodium (Rengasamy *et al.* 2009)."

Wetland salinisation processes

Wetland surface water salinity depends on several factors, including:

- Evaporation from the wetland surface (Barnett *et al.* 2003), including losses from riparian zones (Holland *et al.* 2006)
- The salinity of groundwater and the nature of any interactions between surface water and groundwater (largely driven by differences in watertable and wetland elevation, wetland bed and bank materials and the wetland and groundwater flow geometry (Barnett *et al.* 2003; Jolly *et al.* 2008))
- The connection of the wetland with the river. If well-connected, River Murray water (generally less than 1000 µS/cm; Aldridge *et al.* 2012a) will maintain low salinities that would otherwise increase due to evaporative loss (Barnett *et al.* 2003). This process to some extent depends on any wind-induced mixing with the river (Barnett *et al.* 2003), whether the wetland maintains any through-flow character at pool level, the number of wetlands connections and the difference in elevation across those connections.

The major factors influencing the manner in which groundwater interacts with wetlands, floodplains and the river channel are:

- Depth to the watertable, as shallow watertables increase the likelihood of surface water–groundwater interactions
- Regional groundwater gradients: steep groundwater gradients toward the floodplain will increase the likelihood of interacting and potentially salinising the floodplain and wetlands that lie between the floodplain and the river. Such conditions are commonly observed in association with highland irrigation areas due to increase rates of recharge and high salinity levels
- Aquifer permeability, which affects the flow of groundwater moving to and from surface water features (e.g. wetlands)
- Presence and depth of backwaters (anabranches and wetlands). Deeper backwaters have a higher likelihood of intersecting the watertable and may, depending on the elevation of surface water, change in nature from losing to gaining during a drawdown or to through-flow features when filling and reconnected
- Sill level of backwaters relative to pool level – this influences the degree of connectivity of the wetland to the main channel and therefore the dynamics of surface and groundwater interactions (adapted from Barnett *et al.* (2003); Jolly *et al.* (2012); Souter (2009)).

When hydraulic heads are higher in the surface water within a wetland and the river channel than the surrounding groundwater, low salinity surface water will tend to move laterally from the channel and increase bank storage, with the amount of flow determined by the hydraulic gradient and the aquifer permeability adjacent to the channel (Figure 20; Barnett *et al.* 2003).

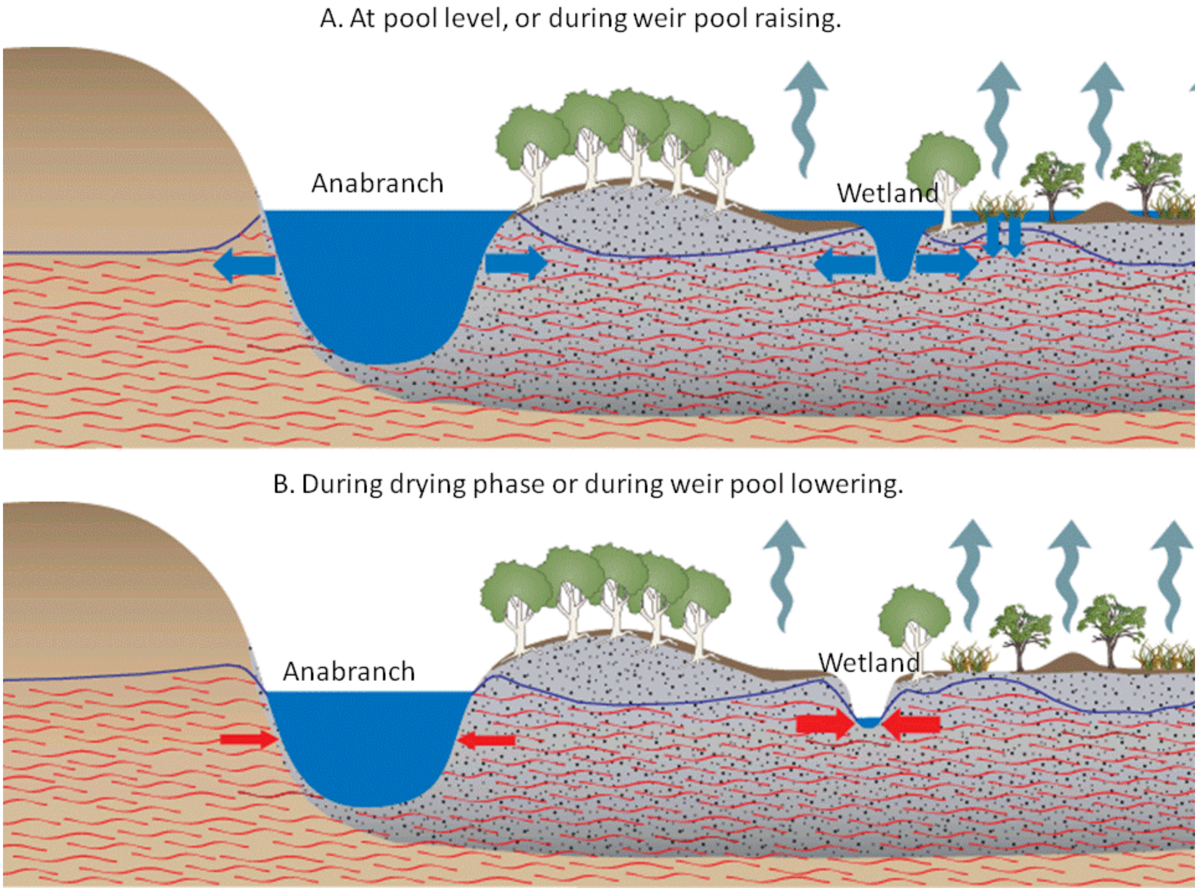

Figure 20: Groundwater flux variation as a function of surface water level. Adapted from Yan and Howe (2008).

- A *When the surface water level exceeds the watertable elevation, water moves from anabranch channels and wetlands and recharges the groundwater.*
- B *With the surface water level lowered (e.g. through a regulator or weir-pool lowering below that of the watertable, there is groundwater discharge back to the surface water.*

Floodplain soil salinity in the root zone and on the soil surface is largely a function of the depth and salinity of groundwater and the frequency and efficiency of flushing flows from the river channel (Barnett *et al.* 2003). As described, by Wallace and Rengasamy (2011) shallow saline watertables can cause salt to accumulate in the root zone of plants. The extent of salinisation that will occur is a function of salinity of the groundwater, depth to groundwater, soil type and rainfall (Rengasamy *et al*. 2009). Capillary action draws groundwater towards the surface. Transpiration by plants and/or evaporation of groundwater through the unsaturated zone leads to salt being left behind and accumulating in soils (SKM 2011).

Conceptualised salintiy responses to water level manipulation

In a static and permanent wetland, the surface water salinity levels will most likely be a function of River Murray water, unless there is significant inflow of saline groundwater from the surrounding landscape (Figure 20). If the wetland is acting as a recharge feature (i.e. losing surface water to groundwater), a freshwater lens of groundwater surrounding the wetland may establish. This may be maintained by the relatively constant River Murray weir pool levels and wetland water levels and thus the hydraulic pressure of the water sitting in the wetland (Figure 21).

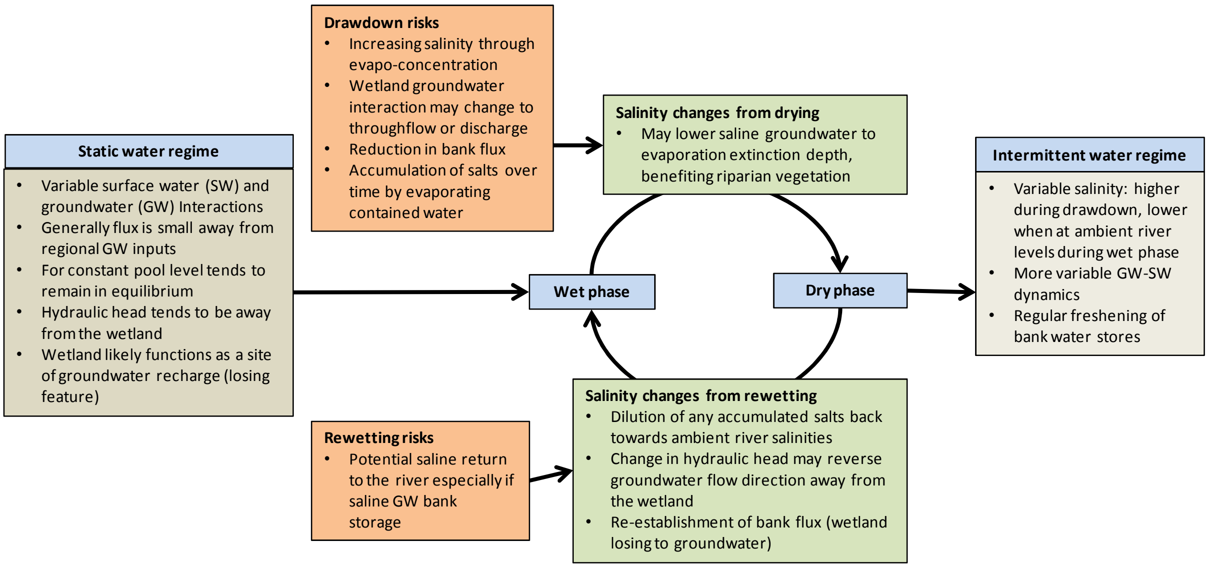

Figure 21: Conceptual model of salinity response to wetting and drying and associated risks in transition from static to intermittent water regimes

Drawdown of wetland water levels, by closing structures and not replacing evaporative losses, will prevent the flux of surface water into the wetlands but may also reduce flux of surface water to the fresh groundwater lens if it sits at pool level or higher. If the wetland has been held at pool level for more than 15 months, this reduction in recharge will most likely benefit river red gums and other amphibious plants accessing bank soil water because it will aerate the sediments and prevent water-logging. However, these plants may become water stressed if the drawdown continues for too long (Vegetation response to water level manipulation section). Wetlands that are partially or completely dried can act as groundwater-flow interceptors because the lowering of the water level within the wetlands by evaporation may induce groundwater flow toward them (i.e. the wetlands may become a discharge feature or gain groundwater).

The critical salinity risks to the ecological response are:

- Increased water salinity due to evapo-concentration or increased groundwater inputs
- Increased soil salinity and sodicity due to retained salt load from saline groundwater and salts left behind by vegetation
- Reduced soil moisture availability for riparian vegetation if the freshwater lens becomes depleted.

When the wetland rewets, any salts that have accumulated will be diluted by incoming river water, trending towards the ambient river salinity levels over time when connected. The hydraulic head may be reversed so that groundwater flows away from the wetland and recharges the bank storages. This may establish or maintain a freshwater lens around the wetland and in so doing, provide soil water to riparian vegetation. If amphibious woody vegetation (e.g. river red gums) have been water stressed this may relieve their stress and lead to measurably improved health.

Increased saline returns to the River Murray when the wetland is filled and reconnects to the river is a significant salinity risk to the River Murray. If the wetland receives significant loads of salt from incoming groundwater during its dry phase, then there may be an increase in the salinity levels in the wetland upon refilling. However, if the wetland does not receive significant loads of salt from incoming groundwater during its dry phase, then there will be no significant increase in the salt balance or salinity levels upon refilling.

It is anticipated that repeated cycles of wetting and drying under an intermittent water regime will lead to more variable surface water salinity, greater variation in the surface water–groundwater dynamics and regular freshening of the wetland bank storage.

Turbidity

Turbidity describes how light is scattered and absorbed by suspended particulate material in the water and is used as a measure of water clarity. Suspended material includes plankton, detritus and inorganic sediments, all of which can impede the penetration of light (Eaton *et al.* 2005). Higher turbidity means lower water clarity and less available light underwater.

Turbidity is a significant driver of biota in various types of Lower River Murray wetlands including permanent lakes, temporary wetlands and floodplains (Souter 2009). Baseline surveys show that the turbidity of permanently inundated Lower River Murray wetlands is highly variable, but typically high, with a mean value of 114 NTU and can reach extreme values (> 400 NTU; Blanch *et al.* 1999a). Highly turbid water can: reduce photosynthesis and primary production at a given water depth; reduce biofilm and aquatic plant growth on the benthos (favouring phytoplankton growth); disrupt zooplankton grazing; modify macroinvertebrate populations; reduce the visual range of sighted organisms, such as fish and water-birds; reduce the survival of eggs and larvae; impede faunal respiration and affect fish behaviour (Davies-Colley and Smith 2001; Henley *et al.* 2000; Jeppensen *et al.* 1999; Johnston 1981; Kemp *et al.* 2005; Schwarz and Hawes 1997). An example of the importance of turbidity is in Lake Alexandrina and Lake Albert, where turbidity is believed to play a principal role in structuring the aquatic ecosystem (Geddes 1984).

The highly turbid nature of wetlands along the SA River Murray may hamper wetland management objectives. Sources of turbidty to these wetlands include:

- Source water: It is not unusual for turbidity in the Lower River Murray to exceed 100 NTU, which is greater than the ANZECC (2000) and South Australian EPA guidelines of 50 NTU for the protection of freshwater ecosystems. Darling River water is typically more turbid than River Murray water. The turbidity of source water will vary depending on antecedent flow conditions and catchment processes.
- Sediment resuspension: Sediment resuspension is the suspension of sediment particles that have previously been deposited on the wetland bed. It occurs when the force required to resuspend sediments, or shear stress, is overcome (Bloesch 1995). Sediment resuspension is particularly important in shallow water bodies and those without abundant aquatic plant communities. Rates of resuspension also depend on sediment type and wetland morphometry.
- Bioturbation: Bioturbation is the mixing of sediments into the water column by animals. Carp are major bioturbators, primarily because of their benthic foraging behaviour,

known as 'mumbling' (Crivelli 1981, as cited in Gilligan and Rayner 2007). However, Gilliagn and Rayner (2007) concluded that there is no consistent relationship between carp and turbidity. Yabbies, water rats and other native fauna also bioturbate, but they are generally considered less problematic than carp. Bioturbation can also have positive effects by promoting oxidation of sediments and enhancing decomposition and nutrient cycling (Mermillod-Blondin 2011).

- Plankton growth**:** Both phytoplankton and zooplankton can significantly contribute to wetland turbidity by scattering and/or absorbing light and reducing light availability for aquatic plants. The calm, warm conditions typical of wetlands of the Lower River Murray provide ideal conditions for phytoplankton growth (Baker *et al.* 2000).

Salinisation can reduce suspended solid concentrations by promoting the coagulation and settling of fine particles (Grace *et al.* 1997; Nielsen *et al.* 2003). In this process salinity neutralises surface particle charge, causing them to collide and coagulate, and precipitate out of solution (Dunlop *et al.* 2005). The salinity at which this occurs depends upon the ionic composition of the water, the type of suspended sediment, and the hydrodynamics of the water body.

Turbidity and alternative ecosystem states

Along with nutrients, turbidity has been linked to the switch of many wetlands from clear, macrophyte-dominated states to phytoplankton (including algae and cyanobacteria) dominated states (Moss and Leah 1982; Scheffer *et al.* 1993). The mechanism for the switch is based on the complex interactions between nutrients, turbidity, phytoplankton and aquatic plants (Scheffer *et al.* 1993). Aquatic plants reduce phytoplankton growth and turbidity by reducing the resuspension of sediments; reducing water column nutrient concentrations; and providing habitat for planktivores (Carpenter and Lodge 1986). Consequently, these wetlands tend to have clearer water and because macrophytes provide habitat and food for a diverse range of aquatic organisms, they typically have greater biodiversity (Scheffer *et al.* 1993). By contrast, highly turbid waters can be almost devoid of submerged aquatic plants, with emergent plants constrained to the wetland fringes by lack of light for growing shoots (Vegetation response to water level manipulation section). High turbidity reduces the growth of submerged aquatic plants due to reduced light availability, which reduces the growth of aquatic plants. This can greatly reduce food resources and habitat complexity, which has implications for ecosystem consumers (Consumer responses to water level manipulation section).

Matsuzaki *et al.* (2009) found invasive carp were acting as possible drivers for catastrophic shifts in ecosystem states. The feeding behaviour of carp damages soft, submerged plants as they are establishing and increases turbidity. The foraging activities of carp may also encourage phytoplankton growth, which has been shown to increase turbidity in upper River Murray wetlands (Gilligan and Rayner 2007). Previous studies have demonstrated that carp exclusion promoted submerged macrophyte growth in shallow areas previously devoid of vegetation (Lougheed *et al*. 2004). A similar study has been conducted by SARDI Aquatic Sciences at Brenda Park, but results are not yet analysed.

Conceptualised turbidity responses to water level manipulation

Reduced turbidity is a management objective in a number of wetland management plans that seek to introduce wetting and drying cycles. The primary management reason for reducing turbidity is to encourage the growth of submerged and emergent aquatic plants, which provide essential habitat and food resources.

Under a static water regime, most River Murray wetlands are highly turbid and contain a low abundance of submerged aquatic vegetation (Figure 22). As the wetland is dried, sediments are exposed and superficially dry out. This leads to water loss, cracking and a change in soil structure such that the sediments become more consolidated, thereby reducing resuspension during the wet phase (Tucker *et al.* 2003; van der Wielen in prep).

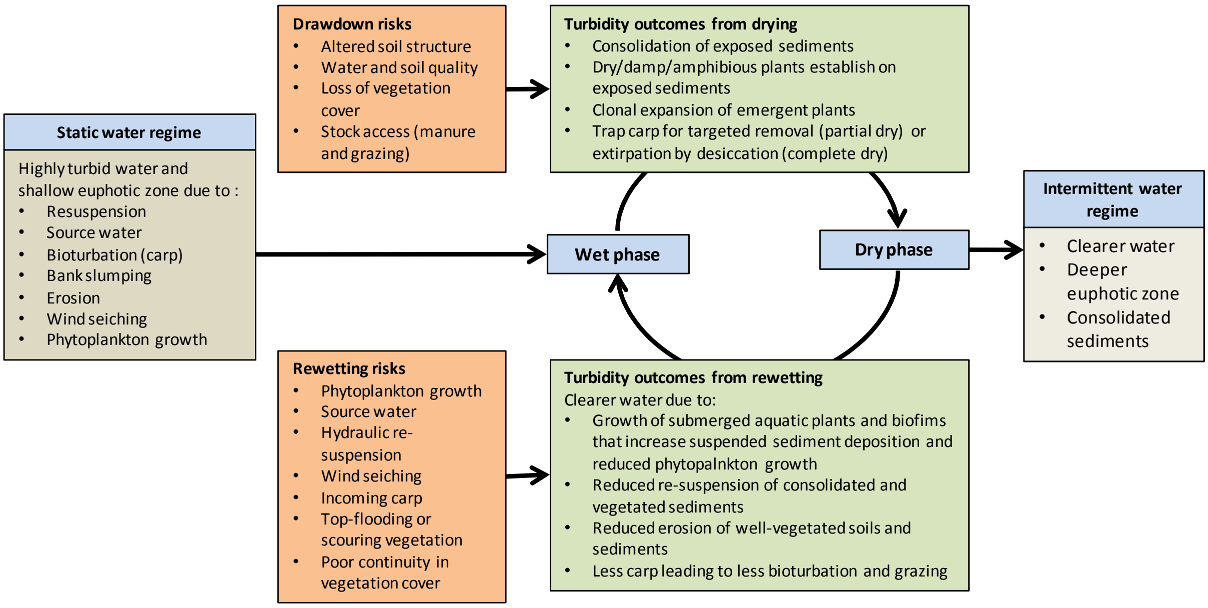

Figure 22: Conceptual model of the processes that control wetland turbidity

Stands of plants established on the wetland bed/fringe during the drawdown phase will stabilise and trap sediments, thereby reducing soil erosion and re-suspension during the wet phase and reducing one source of turbidity. In addition, plants and associated biofilms will compete with phytoplankton for resources, thus reducing phytoplankton biomass and turbidity.

If the wetlands are only partially dried, carp may become highly concentrated causing significant damage to plants that are still inundated and increasing turbidity through bioturbation. Bioturbation by carp may be reduced during critical establishment periods by extirpating resident carp by drying the wetland completely in autumn/winter (after the plants have completed their life cycles and set seed).

The positive effects of wetland drying on turbidity will be constrained by possible adverse alterations in soil structure. Although it is advantageous to consolidate sediments by drying, excessive moisture loss from wetland sediments can lead to very deep cracks or other physico-chemical changes in soil structure (e.g. mineralisation) that will not be reversed by re-wetting and may impede other wetland functions such as plant germination and bioturbation (e.g. deep cracks in wetlands below Lock 1 that have not refilled; K. Mason, Department of Environment, Water and Natural Resources, pers. comm.).

Upon refilling, fish screens on wetland inlets only prevent large fish (including carp) entering the wetland upon refilling. Juvenile carp will enter and are likely to have an adverse effect on resuspension and germinating or reshooting submerged and amphibious plants. If the aquatic plant beds are well-established, the increases in turbidity due to bioturbation should be significantly lower, provided the plants are able to maintain their cover in the presence of carp.

Biogeochemical cycles

Biogeochemical cycles are the ways in which matter (e.g. macro and micronutrients, organic material) changes form. For example, the decomposition of plant and algal detritus by sediment microflora (i.e. bacteria, fungi) utilises a range of biogeochemical pathways. The particular pathways will depend on the characteristics of the given wetland, including sediment type and hydrology, and will have implications for surface water quality, in particular the availability and form of nutrients and metals. This has implications for wetland biota. If concentrations are too low then productivity may be limited, but high concentrations may be beyond physiological tolerances and thus be toxic to biota. In addition, elevated nutrient concentrations can favour phytoplankton and thus limit macrophyte abundance and diversity.

Nutrient cycling

Nutrients are substances that are metabolised by organisms to give energy and build tissue (Wetzel 2001). These include nitrogen, phosphorus, sulfur, potassium, magnesium and calcium. Of these, phosphorus and nitrogen are often the nutrients that limit the growth of biota with the remainder usually found in amounts that exceed the requirements of autotrophic organisms (Kalff 2002).

Nitrogen occurs in various oxidised and reduced forms with the availability of each form often regulating riverine productivity (Shafron *et al*. 1990). As with all nutrients, the nitrogen cycle in aquatic systems is complex, with continuous cycling between sediments, dead organic matter, the water column and the various components of the aquatic food-web. Within a wetland, nitrogen is cycled through the following processes:

- Nitrogen fixation: an enzyme-catalysed process, through which N_2 is converted to ammonium (NH_4) and organic forms of nitrogen (Kalff 2002).
- Ammonification: NH_4 is the end product of decomposition of particulate and dissolved dead organic matter by heterotrophic bacteria. This involves the deamination of proteins, amino acids, urea and other nitrogenous organic compounds (Wetzel 2001).
- Nitrate assimilation: NO_3 must be reduced to NH_4 before it can be assimilated by plants (Wetzel 2001). Upon reduction to NH_4, it may be incorporated into the microbial/algal protoplasm (Kalff 2002).
- Nitrification: the biological oxidation of organic and inorganic nitrogenous compounds from a reduced state, which begins with the conversion of NH_4 to NO_2, which then proceeds further to NO_3 (Kalff 2002; Wetzel 2001).
- Denitrification: a bacterially mediated process of reducing oxidised nitrogen anions (NO_X), with the concomitant oxidation of organic matter. Thus denitrification plays a significant role in organic matter decomposition. Oxidised nitrogen is firstly reduced to nitrous oxides (NO, N_2O) and then to N_2 gas (Wetzel 2001).

Phosphorus cycling in the aquatic environment is complex due to the transformation of phosphorus between organic and inorganic fractions in both soluble and insoluble forms. More than 90% of the phosphorus in freshwater ecosystems consists of organic phosphorus (Wetzel 2001). Dissolved inorganic phosphate, more commonly referred to as orthophosphate or filterable reactive phosphorus (FRP), is the most biologically available form of phosphorus, which is derived from the breakdown of dissolved organic phosphorus and also release from sediments by biochemical processes (Shafron *et al*. 1990).

Phosphorus enters aquatic systems from the terrestrial environment and from direct atmospheric deposition on the water surface (Kalff 2002). Contributions from the catchment normally dominate total inputs except where catchments are very small and composed of soils with low phosphorus concentrations (Kalff 2002). Uptake by macrophytes, particularly rooted vascular plants, is generally much less than by attached algae and other microbes (Wetzel 2001). Phosphorus can be released from biota by excretion in inorganic and organic forms from living microbiota or as the organisms senesce, die and lyse.

Under oxidised conditions, orthophosphates strongly adsorb to sediments due to the oxidation of ferrous sulphides into amorphous ferric oxyhydroxides, which have a high affinity for phosphorus (Baldwin 1996; De Groot and Fabre 1993; De Groot and Van Wijck 1993). This oxidised surface layer may also prevent diffusion of phosphorus and reduced mineral ions from deeper sediments (Hutchinson 1957; Moss 1988), including those from the decomposition of sedimented organic particles. However, if oxygen levels are depleted within the sediment, iron oxyhydroxides are reduced, causing the dissolution of iron and phosphate from the sediment surface. Release of phosphorus from reduced sediments has been shown to be far greater than from oxidised sediments in many lakes throughout the world (Jensen and Andersen 1992; Lennox 1984; Marsden 1989; Mortimer 1941, 1942; Sondergaard *et al.* 1993; Stephen *et al.* 1997).

Acid Sulfate Soils

Cycling of sulfur will occur in any sediments that contain sulfur and undergo a wetting and drying cycle. This is not problematic unless the soils contain large amounts of sulfur where sulfate reduction is a significant catabolic pathway and become acid sulfate soils (ASS). Acid sulfate soils are defined as soils or sediments that contain (or once contained) high levels of reduced inorganic sulfur (mostly as sulfide, elemental sulfur, or both) and when exposed to oxygen, the soils or sediments undergo a chemical reaction that produces acid (EPHC and NRMMC 2011). In order for ASS to form there needs to be supplies of iron minerals, organic matter and sulfate as well as reducing conditions in the sediment which will support sulfate-reducing bacteria to convert the sulfate ions to sulfides such as ferrous sulfide (Fitzpatrick *et al.* 2008). In recent years ASS have been identified in many Australian inland aquatic ecosystems, including numerous wetlands throughout the Murray-Darling Basin. A number of associated risks posed by ASS have also been identified (EPHC and NRMMC 2011) including:

- Acidification: generation of acid via a series of complex oxidation reactions when ASS are exposed to oxygen. If the amount of acidity produced by this oxidation process is greater than the system's ability to absorb that acidity (the acid neutralising capacity) the pH of the system falls.
- Deoxygenation: some ecosystems containing ASS have high capacity to neutralise acid and may not acidify. However, ASS oxidation consumes oxygen and can deoxygenate the water resulting in extreme anoxia events that lead to mortality of

aquatic organisms (e.g. fish kills). Deoxygenation is most likely to occur if monosulfidic materials (formerly monosulfidic black oozes), are physically disturbed and distributed throughout a water column.

- Release of metals and metalloids: Oxidation of sulfidic materials may lead to heavy metals (such as cadmium and lead) and metalloids (such as arsenic) becoming more available in the environment. Once freely available in the environment they can be directly incorporated into living tissue and potentially enter the food chain. Dissolved aluminium, the most common and harmful metal released, is toxic to many aquatic plants and fish (ANZECC 2000). It can be released from clays that are broken down under acidic conditions. Metal flocculants may also form, which can be fatal or cause injury to organisms with gills.

The ASS materials present in wetlands can change from being sulfidic (relatively benign) to sulfuric (generating acid) and back again depending on environmental conditions (Figure 23). When wet, the ASS is under reducing conditions and the ASS materials will tend towards the sulfidic state. When exposed to air (dry or disturbed) and under oxidising conditions, the ASS materials will tend towards the sulfuric state. Acidified soil thus represents an environmental hazard and the potential for acidification of the water body increases as the area of sulfuric material increases. Flora and fauna can be affected by acidification directly if the pH drops to below their tolerance, or indirectly via oxidation products such as heavy metals. ASS that is kept wet is likely to remain in a reducing state and does not represent an environmental hazard.

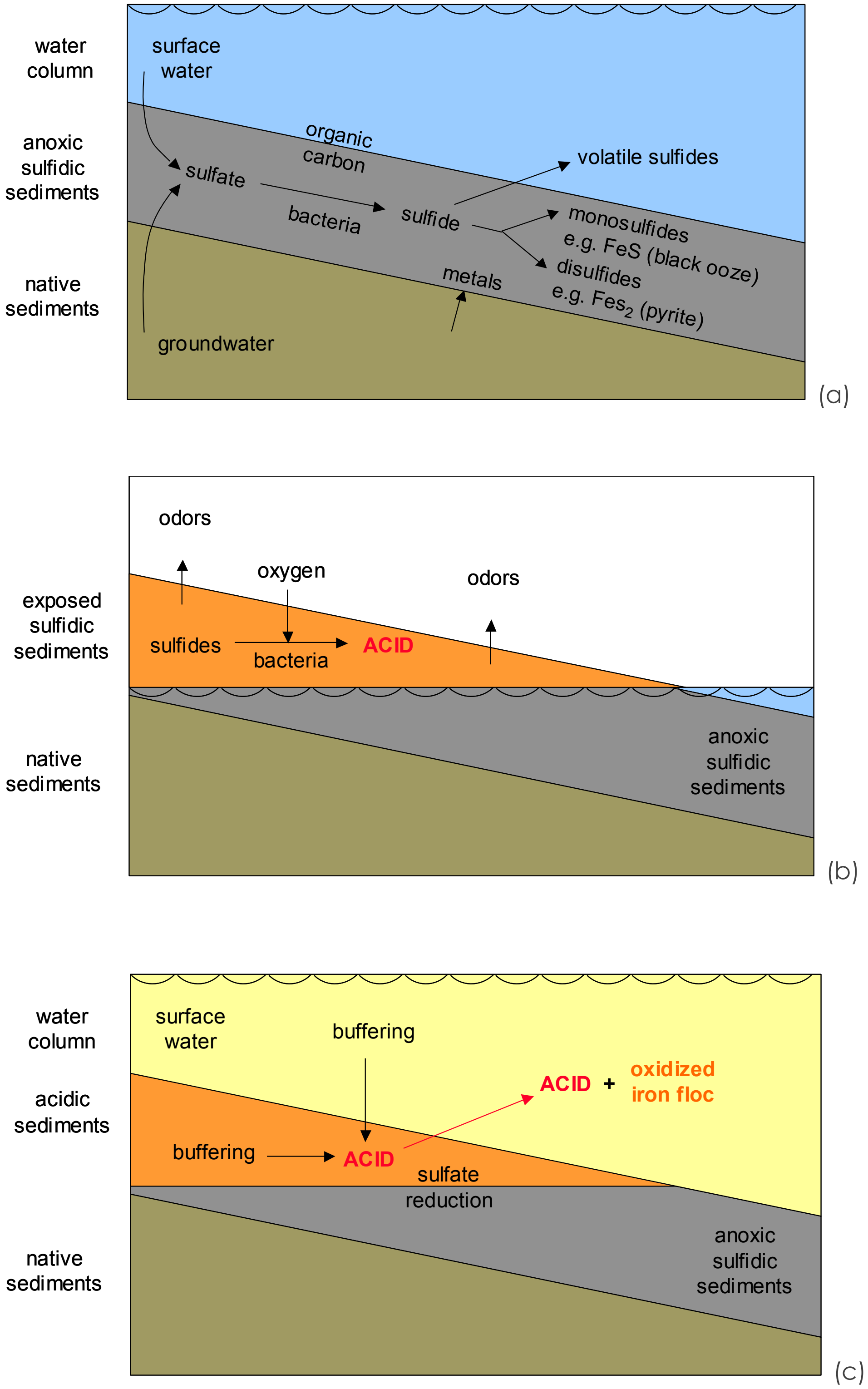

Figure 23: **Schema of acid sulfate soils (ASS) under different water regimes. Biological formation of ASS through sulfate reduction when submerged (a), generation of sulfuric acid via sulfide oxidation when ASS exposed (b) and flushing of acids from ASS into the water the water column when exposed ASS are rewetted (c). Source: Baldwin (2009).**

Summary

This report presents a conceptual understanding of how wetlands of the Lower River Murray that have been permanently inundated due to regulation may respond to future water level manipulations. Overall, it is anticipated that the RRP will to restore a gradient in hydrological regimes across the landscape of the Lower River Murray. In doing so, it is anticipated that wetlands will have improved diversity and abundance of vegetation, which will provide increased habitat and food for consumer groups, such as frogs, fish and water-birds. Thus the diversity and abundance of consumer groups is also expected to improve. Whilst, much of the information presented in this report describes general ecological responses anticipated from water level manipulations, it is not anticipated that the ecological response to water level manipulations will be the same for each wetland. Indeed, biological diversity needs to be managed at the landscape scale and not the wetland scale.

This conceptual understanding was developed through a combination of literature, observations and expert opinion. It forms a basis upon which the monitoring and evaluation program will provide evidence for the actual response to water level manipulations. Details on the technical design of the monitoring and evaluation program are provided in DEWNR (2012).

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Appendix 1 – Designing wetting and drying cycles to optimise ecological responses

Based on workshops with relevant experts, the following information was generated for designing wetting and drying cycles for ecological outcomes. However, it is important to note, that consideration needs to be given for the diversity of wetland types across the landscape in order to maximise landscape biodiversity.

Vegetation

Overall, the diversity and abundance of wetland vegetation will be optimised if the wetland hydrograph seeks to:

- Drawdown the wetland in late summer/early autumn to encourage germination of dry, damp and amphibious plants (< 20 mm per day is optimal, < 0.1 m per day is acceptable)
- Drawdown to at least 0.1 m lower than the emergent plant band at least one in every two years and to the minimum target elevation at least one in every four years to stimulate recruitment, growth and maintain aquatic vegetation to the extent of the euphotic depth
- Drawdown to 0.5 m below the elevation at which target river red gums occur at least every 15 months to provide root aeration
- Refill quickly enough so that the maximum drawdown period is 6 months (< 20 mm per day is optimal, < 0.1 m per day is acceptable although faster refill may limit the survival of certain amphibious plants)
- Refill the wetland to cover terrestrial weeds before seed-set, to reduce fresh seed entering the seed bank, or at least 9-12 months after recruitment during the current drawdown
- Refill the wetland in late winter/spring to encourage growth of amphibious and submerged plants and provide soil water to amphibious woody-riparian and floodplain plants (< 0.1 m per day is optimal)
- Increase the water level to the terrestrial edge of the target floodplain area at least one in every ten years to stimulate recruitment, growth and maintain wetland vegetation to the terrestrial edge of the floodplain
- Vary the maximum and minimum depths of water across successive years to encourage aquatic plant diversity and overlap boundaries between different functional groups
- Implement complementary works to control pest plants such as willows, *Xanthium* spp. and *Juncus acutus*. Refilling before terrestrial weeds have set seed will reduce seed build up in the wetland, although most weeds have highly dispersive seed that blows in or is carried in by water.

The exact sequencing and extent of wetting and drying cycles should be informed by the specific targets for the wetland, as defined by the relevant management plan and from an evaluation of vegetation monitoring data.

Other complimentary actions, such as control of Carp or grazing management, may also be needed to optimise vegetation outcomes.

Frogs

In general, the ideal wetland habitat for a range of frogs would have inundated margins from late winter to late summer/early autumn and diverse and complex aquatic vegetation, including:

- Open, inundated, reed beds that allow tadpoles to move around whilst providing cover
- freshly inundated riparian grasses and forbs
- submerged and floating plants or algal mats that are dense enough to support adult frogs above the water surface
- open amphibious vegetation, river red gums, black box, lignum, emergent plants, dead reeds, flooded grasses/herbs and other plant structures that frogs can use to perch above the water surface.

Wetting and drying at a wetland scale will enable frog populations to be maintained but weir pool raising or floods are required to inundate riparian vegetation and support dispersal and to optimise recruitment and population connectivity.

Fish communities

Management objectives in many plans include control of pest species such as Carp, which requires complete wetland drying to kill resident Carp. However Carp are a highly opportunistic species that can move and spawn across a wide range of conditions. Furthermore, all fish contained the drying wetland will be lost, not just the alien target species.

Overall, the diversity and abundance of native freshwater fish are most likely to be optimised if the wetland hydrograph seeks to:

- Keep the wetland structures open from August to at least April to allow an extended period of connectivity between wetlands and the main channel
- Open structures when there is a simultaneous rise in flow rate within the main river channel to enhance cues to move into wetlands
- Begin drawdown, If possible, when there is a simultaneous drop in river level within the main channel, then briefly open the structures to allow fish to exit the wetland, before becoming trapped in a wetland prior to a complete dry. It is expected that native fish will leave earlier than carp
- Avoid drying a suite of adjacent wetlands at the same time and ensure that permanent refuge habitat is available for fish escaping a drawdown
- Avoid complete drawdown in wetlands that contain rare species such as Murray hardyhead and Southern purple-spotted gudgeons, although fluctuating water levels may optimise their habitat, therefore careful implementation of partial drawdowns may be beneficial

- Base water quality triggers around the tolerances of the larval stages of fish (i.e. <5000 µS/cm; Ye *et al.* 2010) to ensure that fish recruitment is supported in addition to the survival of more tolerant adults.

Water-birds

Overall, the diversity and abundance of water-birds are most likely to be optimised if a range of wetland habitats is provided at the landscape scale. Individual wetland hydrographs should seek to fill gaps in habitats provided within local areas such as:

- Expose mudflats near the wetland edge and if possible fringing reed beds. Keep these mudflats moist for several weeks in spring/summer to provide foraging habitat for waders, shorebirds, rails and crakes
- Promote 'islands' of reeds and trees in some wetlands to increase the extent of edge habitat that is surrounded by water and is thus protected from terrestrial predators
- Maintain water under nesting trees or shrubs long enough to allow nestlings of flood-dependent breeders to fledge
- Fluctuate water levels in wetlands to support complex and diverse edge habitats that provide close-proximity nesting and foraging habitats
- Provide water to support dense and contiguous stands of reeds, lignum and river red gums in some wetlands to support breeding of ibis and cormorants, especially where these are known to breed or have bred historically, as they have a level of site fidelity
- Promote large stands of *Phragmites australis,* or *Typha* spp., or both in some wetlands to support Australasian and Little bitterns, rails and crakes
- For a complex water-bird community, implement a water regime that provides a mosaic of shallow margins, deep water and diverse, interconnected vegetation assemblages.

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